Monitoring the Impact of Scott Base in Antarctica:

A Recent Evaluation of Wastewater, Water and Soil Quality at Pram Point, Ross Island.

Thomas Mervyn Williams

A thesis submitted in partial fulfilment of the requirements for the Degree of Master of Science in Environmental Science

at the University of Canterbury

University of Canterbury

2012

"…although scientific research in Antarctica is of paramount importance in addressing climatic and environmental challenges, there is no doubt that the value of Antarctica for science should be weighed against the environmental impact of scientific work and its logistic support."

Bargagli (2005)

Abstract

Antarctica is widely perceived as the most untouched continent on Earth. However, increasing anthropogenic presence in Antarctica is creating continual pressure on the pristine environment. To protect the Antarctic environment, monitoring and reporting procedures must be a priority for Nations wishing to conduct research on the continent. A significant contributor to environmental degradation is poor waste management and waste disposal, in particular the discharge of sewage and wastewater into the marine environment. This study provides information on the potential impact of Scott Base wastewater on the local marine environment, which can be used to improve operating systems and as a tool to ultimately reduce the environmental footprint of the base.

This study investigated the characteristics of the sewage from Scott Base, Antarctica, and the water quality within the wastewater discharge plume beneath the sea ice. Results from seawater analysis were then compared with Redvers (2000) to give an indication of how contamination levels have changed over the last decade. Results show that in the vicinity of the wastewater outfall, seawater samples contained no faecal coliforms or *Escherichia coli*. Nitrate-Nitrogen (NO_3^-- N) concentrations were recorded up to 1.1 mg/L, while phosphate (PO_4^{3-}) ranged from 0.28 to 0.45 mg/L. The pH ranged from 7.84 to 7.92. Dissolved oxygen ranged between 10.05 and 13.02 mg/L, and conductivity between 48.4 to 55.2 mS/cm. Concentrations of copper (Cu), manganese (Mn), and zinc (Zn) were greatest at sites within 30m of the outfall. Iron (Fe) and nickel (Ni) were detected at most sites within the plume. The general spatial extent of the plume is now approximately 50m long-shore, and 30m offshore. Compared to Redvers (2000) the current plume is more localised, with a reduction of approximately 100-125m long-shore and 10-20m offshore. Faecal coliforms have reduced to no longer be detected within the plume, while dissolved oxygen and total organic carbon concentrations in the plume have increased, and conductivity has decreased.

Soils surrounding Scott Base were also investigated as a potential source of trace metal (As, Cd, Cr, Cu, Fe, Mn, Ni, Pb and Zn) discharge to the marine environment. Total recoverable soil metal concentrations and readily leachable metal concentrations were measured. Sites were selected on the basis of being potential sources of contamination (fuel storage areas or refuelling areas), or potentially effecting the marine environment (shorelines). Results show that the greatest concentrations of metals and readily leachable metals were found primarily along the shoreline, and not necessarily where contamination related to current base activities is likely to occur. The results indicate that historic contamination may still persist in the terrestrial environment, and has the potential to be transported into the marine environment.

Domestic and drinking water samples were analysed for trace metals and nutrients to ensure that contaminants from the wastewater outfall were not entering the reverse osmosis plant. Results show that the drinking water meets New Zealand Drinking Water Standards (Ministry of Health, 2008).

Acknowledgements

I am extremely grateful to the Christchurch City Council, Antarctica New Zealand and Gateway Antarctica for the *Christchurch City Council Scholarship* and logistical support for field work.

To Dr. Sally Gaw, Prof. Jenny Webster-Brown and Prof. Bryan Storey, thank you for your continual support throughout this study, and broadening my knowledge in a research field that was reasonably unfamiliar at first. Many thanks to Rob Stainthorpe for his perseverance in trace metal analysis by ICP-MS.

Thanks are also due to Associate Prof. Wendy Lawson from the University of Canterbury and Jana Newman from Antarctica New Zealand for encouraging the initial beginnings of the project. Also, Prof. Clive Evans, University of Auckland, for 'on ice' guidance and mentoring during field work.

Thank you to the many staff at Scott Base for helping in one way or another, the Engineers, in particular Darren Hopkins who repaired gear and cleared snow, and the Science Technicians, Nita Smith and Victoria Landgraf. Many thanks must go to those that lent a hand drilling ice and collecting samples, Paula Gentle, Erin Neufield, Nimrod Kiss and Jürgen Kolb.

To my family and friends, thank you for your continual support, not only during my studies, but in all aspects of my endeavours. I look forward to sharing the next adventure with you all!

Table of Contents

List of Figures

List of Tables

Abbreviations

DO	Dissolved Oxygen
NO_3^-- N	Nitrate - Nitrogen
PO_4^{3-}	Phosphate
RO	Reverse Osmosis
RO1	Water used for domestic purposes
RO2	Drinking water
STP	Treated Wastewater
TOC	Total Organic Carbon
WWTP	Wastewater Treatment Plant

1 Introduction

1.1 Anthropogenic Impacts in Antarctica

Antarctica is often perceived to be an isolated and desolate place, the last great wilderness untouched by human influences. However, since exploration began in the heroic age, the Antarctic environment has become increasingly degraded (Bargagli, 2008). Environmental degradation increased significantly during the International Geophysical Year (IGY) in 1957-1958 when 55 stations were established on the continent and Sub-Antarctic islands. The majority of permanent and semi-permanent bases are situated in coastal environments, or on ice-free terrain (Gröndahl *et al.*, 2009), and currently some 4000 to 5000 people occupy these stations and semi-permanent field camps during the summer, and 1000 during the winter (Bargagli, 2008; Elliott, 2005; Gröndahl *et al.*, 2009). Added to this is the pressure of fishing activities and tourism of the last 50 years (Bargagli, 2008), with 37,550 tourists visiting Antarctica during the austral summer of 2006/07 and being concentrated mainly in coastal areas (Gröndahl *et al.*, 2009).

Two Antarctic bases are present on the Hut Point Peninsula, Ross Island (Figure 1). The United States of America's McMurdo Station (77°50'S, 166°40'E) was built at Hut Point in 1955 (Figure 2), and is occupied by 250 people during the winter and up to 1200 in the summer (Crockett, 1997). The neighbouring Scott Base (77°51'S, 166°46'E) was established by New Zealand on Pram Point in 1957 (Figure 2) as part of the Commonwealth Trans Antarctic Expedition and the IGY. The total occupancy of Scott Base reaches up to 100 during the summer and is reduced to approximately 15 over the winter (Anderson and Chague-Goff, 1996; Elliott, 2005; Miller *et al.*, 1999).

One of the most significant impacts of research facilities in Antarctica is the discharge of sewage and wastewater into the environment. Historically, 'out of sight disposal' of human waste was practised, with waste either discharged at sea, or buried in ice pits to minimise human contact and for aesthetic reasons (Smith and Riddle, 2009). In the 1960s, increasing environmental awareness led to the implementation of a Code of Conduct for Antarctic Expedition and Station Activities (Bleasel, 1989). This Code stated that coastal bases should macerate wastewater, and discharge into deep waters. With regards to inland bases, it states wastes should be removed to supporting bases or ships, and if this method was deemed impractical, deep ice pits were to be used (Bleasel, 1989). However, few bases adopted the recommendations within the Code, and untreated wastewater was often discharged directly into the near shore sea-ice or intertidal zones (Redvers, 2000). Further demand for environmental protection in Antarctica during the late 1980s prompted the development of the Protocol on Environmental Protection to the Antarctic Treaty (also called The Madrid Protocol). This rigorous legislation was adopted in 1998 to protect the Antarctic environment, and is includes mandatory obligations to monitor the impacts of wastewater discharge.

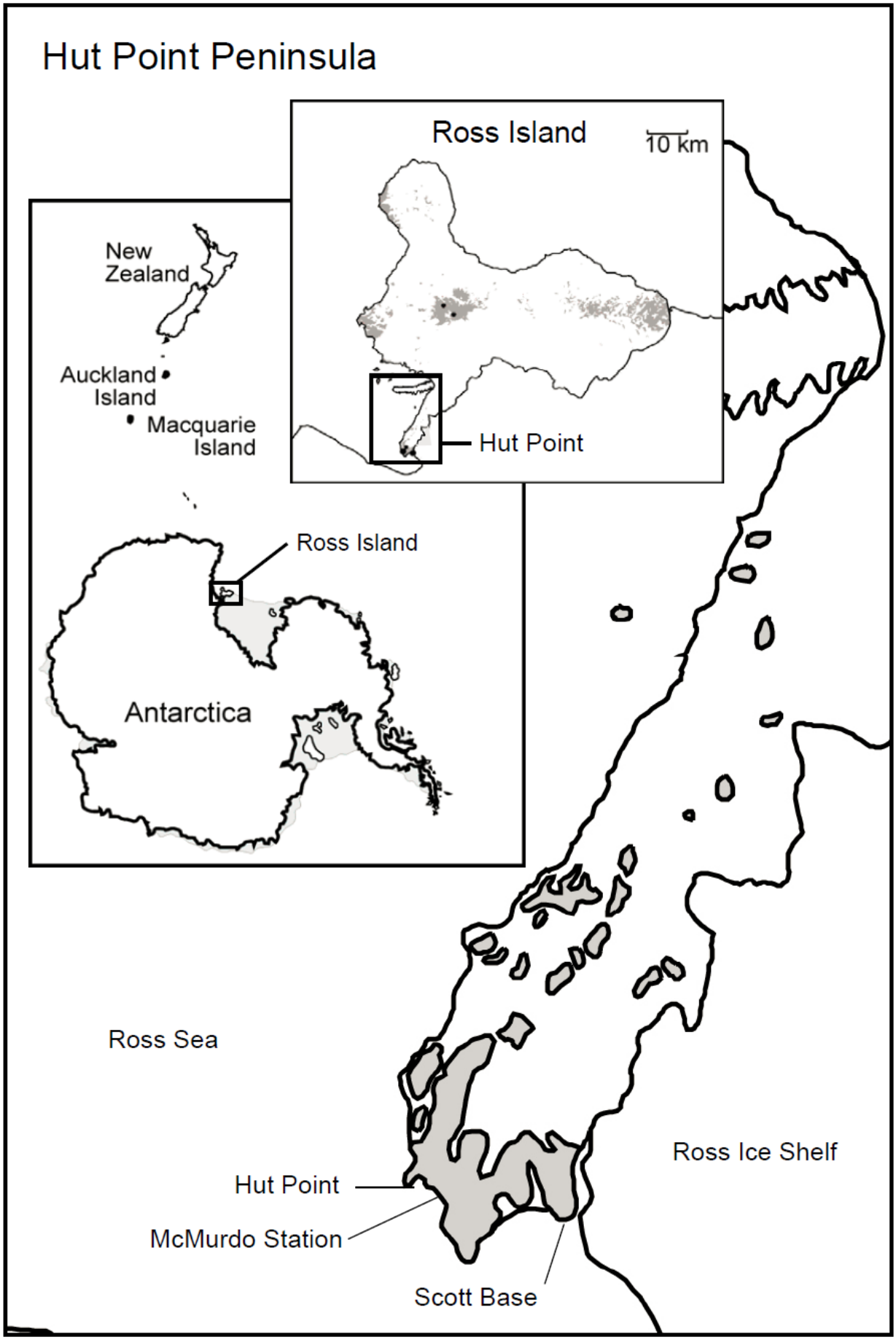


Figure 1. Location of Scott Base and McMurdo Station, Ross Island, Antarctica (Antarctica New Zealand, 2001).

Figure 2. McMurdo Station (top) and Scott Base (bottom), Ross Island, Antarctica.

1.2 The Antarctic Treaty System

The Antarctic Treaty System (ATS) incorporates agreements designed to regulate relations among the states in Antarctica. At the core of the ATS is the Antarctic Treaty, which primarily ensures *"in the interests of all mankind that Antarctica shall continue forever to be used exclusively for peaceful purposes and shall not become the scene of object of international discord"* (Secretariant of the Antarctic Treaty, 1959, pg. 1). Additionally, Recommendations adopted at Consultative Meetings provide further protection to Antarctic environment. The most pertinent of the Recommendations made is the Protocol on Environmental Protection to the Antarctic Treaty (the Madrid Protocol). This agreement essentially prevents development and provides protection for the Antarctic environment, and will be discussed further in the following section.

1.2.1 Requirements under the Madrid Protocol

The Madrid Protocol was agreed upon by Treaty Nations in 1991, and adopted in 1998. The Treaty parties were henceforth committed to the *"...protection of the Antarctic environment and dependant and associated ecosystems and the intrinsic value of Antarctica, including its wilderness and aesthetic values...*" (Madrid Protocol, 1991, article 3(1)). Since the adoption of the Madrid Protocol, significant efforts have been made to improve existing, and to reduce future anthropogenic impacts on the Antarctic environment. The Madrid Protocol bound all signatories to abide by environmental obligations and standards under five technical annexes. These annexes include marine pollution, waste management, conservation of flora and fauna, protected areas, and environmental impact assessments (Madrid Protocol, 1991).

Generally, all activities conducted in Antarctica require environmental impact assessments, as outlined under Annex I of the Madrid Protocol. Three levels of assessments are applied to activities with a variety of probable impacts: Preliminary Assessments (PA), Initial Environmental Evaluations (IEE) and Comprehensive Environmental Evaluation (CEE). More specifically, provisions concerning sewage practices in Antarctica are detailed in Annexes III and IV of the Madrid Protocol. It is stated within the Annexes that sewage and domestic liquid waste must not be discharged on to sea ice, ice shelves or grounded ice-sheets, but may be discharged into the marine environment provided that the conditions are favourable for initial dilution and rapid dispersal. Also, large volumes of sewage, generated by an average summer station population of 30 people or greater, should be treated by maceration at minimum prior to discharge. The by-products of sewage treatment by biological processes may also be disposed of into the sea if there are no adverse affects on the marine environment (Madrid Protocol, 1991).

The responses of signatory nations to the Madrid Protocol have varied, yet the majority of countries have used the opportunity to introduce more advanced sewage treatment methods (Connor, 2008). Despite many counties meeting their obligations under the Madrid Protocol to reduce environmental impact from sewage discharge, 37% of permanent stations and 69% of summer stations lack any form of sewage

treatment (Gröndahl *et al.*, 2009). This is of concern as even the minimal discharge of untreated waste from small, summer only stations can have a substantial impact on the marine environment (Crockett and White, 2003; Stark *et al.*, 2003a).

1.2.2 Environmental Monitoring of Sewage and Wastewater Impacts

Increasing anthropogenic occupation is resulting in greater pressure on the environment, which needs to be strictly controlled and monitored. To protect the Antarctic environment, environmental monitoring and reporting procedures must be established. Monitoring primarily determines if a proposed activity is being conducted in accordance with the CEE, and is a legal obligation for signatory nations under the Antarctic Treaty System (Dingwall, 1998).

The Environmental Protocol (1991) states environmental principals in Article 3 (2) of the preamble, which include;

"activities in the Antarctic Treaty area shall be planned and conducted so as to avoid:

- *significant adverse effects on…water quality*
- *significant changes in the…marine environments*
- *detrimental changes in the distribution, abundance or productivity of species or populations of species of fauna and flora"*

The principals also state that;

> *"Regular and effective monitoring shall take place to allow assessment of the impacts of ongoing activities, including the verification of predicted impacts" and "regular and effective monitoring shall take place to facilitate early detection of the possible unforeseen effects of activities carried on both within and outside the Antarctica Treaty area on the Antarctic environment and dependant and associated ecosystems."*

Systematic monitoring will establish long-term records in order to better understand anthropogenic impacts on the Antarctic environment (Tin *et al.*, 2008). Monitoring of station activities includes fuel consumption, waste incineration, spills, station area and wastewater production (Geochemical and Environmental Research Group, 2000). Specifically for sewage and wastewater impacts, monitoring provides quantitative and qualitative data that measures toxic substances in wastewater streams, and the effectiveness of treatment (Crockett, 1997). A range of parameters can be monitored in sewage and wastewater to investigate the potential for contamination of the water column and marine sediments. For monitoring sewage and wastewater in Antarctica, the Geochemical and Environmental Research Group from Texas A&M University, USA, developed a technical handbook of standard techniques. The prescribed indicators of wastewater included total suspended solids (TSS), dissolved oxygen (DO), biological oxidation demand (BOD), chemical oxidation demand (COD), acidity, conductivity, nutrients,

temperature, and coliform bacteria (GERG, 2000). Other indicators used worldwide include heavy metal concentrations (Ribeiro *et al.*, 2011), molecular biological indicators (Negri *et al.*, 2006; Stark *et al.*, 2003a) and geochemical indicators (Montone *et al.*, 2010). Studies have also highlighted the need to use indicator species to measure contamination, with Cunningham *et al.* (2005) revealing that benthic diatom communities are useful for monitoring future metal contamination. Benthic organisms are well suited to environmental impact studies as they have restricted mobility and are situated where organic matter and pollutants accumulate (Bargagli, 2006). In an extensive study conducted by Conlan *et al.* (2004), 10 years of change in the benthic communities along the coast of McMurdo Sound was observed prior to the installation of the WWTP at McMurdo Station. Conlan *et al.* (2010) found that since sewage treatment commenced at McMurdo, the composition of the benthic community within Winter Quarters Bay is altering in a positive way. The abundance of sewage associated polychaetes is declining, with those found at more pristine sites increasing.

The Environmental Protocol recognizes that long-term, prolonged observations are essential for managing anthropogenic impact in the unique Antarctic environment. Hughes (2010) emphasised that monitoring programmes of a high standard should be undertaken for all stations with large infrastructure. Monitoring of sewage and wastewater should include an assessment of the physical disruption of both marine and terrestrial habitats, and record the levels of pollutants. This will allow assessment of the impacts upon the entire range of biological groups in the local ecosystem. Monitoring in Antarctica also requires specific programs as some contaminants have been found to vary greatly even within sites of similar characteristics (Curtosi *et al.*, 2010), yet one approach to assessing anthropogenic impact is to compare data collected with pristine and remote areas. The benefit of monitoring in Antarctica is the same type of data can be readily compared to background levels with relative ease as vast areas of Antarctica are not visited (Sanchez - Hernandez, 2000). However, factors that could limit monitoring include a lack of monitoring expertise, limited access to complicated technical resources, or less emphasis on routine monitoring compared to field research priorities.

To encourage monitoring, in 1994 the Antarctic Treaty Parties requested the Council of Managers of National Antarctic Programs (COMNAP) and Scientific Committee on Antarctica Research (SCAR) to develop a method of environmental monitoring in Antarctica that had a strong emphasis on being scientifically sound, and cost effective. The subsequent report presented in 1996 titled, 'Monitoring of Environmental Impacts from Science and Operations in Antarctica' included the 'Antarctic Environmental Monitoring Handbook' to guide scientific protocols for environmental monitoring (GERG, 2000). The purpose of the handbook is *"to provide guidance for the measurement of a first tier of indicators that are most relevant to the monitoring impacts due to scientific stations and the associated support activities"* (GERG, 2000).

COMNAP and SCAR recognize the need to review and revise monitoring methods as National Antarctic Programs collect data, gain experience, and as technologies are advanced. It is stated that *"the manual*

should be updated and amended on an as needed basis" and that *"methods can be revised or replaced when sufficient information is available to justify revision"* (GERG, 2000). In 2005 a working paper from the Antarctic Treaty Consultative Meeting XXXVIII has provided an updated titled "Practical Guidelines for Developing and Designing Environmental Monitoring Programmes in Antarctica" (COMNAP, 2005), and in 2010 the Committee for Environmental Protection endorsed the guidelines. If the handbook is widely adopted, and information effectively shared, this will most certainly achieve a wider perspective on continental scale anthropogenic impact.

To promote the use of the handbook, the working paper also developed a three-step model to designing and developing an environmental programme, or adjust and improve existing monitoring (Figure 3). The development and widespread promotion of these tools are essential for establishing comprehensive long term monitoring programmes. This is important as the need for long term monitoring data is crucial for making future decisions on how to treat wastewater, the design of outfalls, and also the likely impacts of wastewater disposal on Antarctic flora and fauna (Tin *et al.*, 2008).

Sharing information is also essential to the success of environmental monitoring programmes. At present, information on the extent and impacts of sewage pollution is only publically available for several research stations (Aronson *et al.*, 2011; Gröndahl *et al.*, 2009). Hughes (2010) reported that out of 75 research stations on the continent, very few publish their environmental monitoring data and under the Antarctic Treaty. Signatory nations are required to freely exchange information of their monitoring work under Article VII of the Treaty, yet only three of the 28 nations complied and submitted information for the 2008/2009 seasons.

Many National Antarctic Programs have plans to monitor, or have already been monitoring the effects of wastewater discharges. The USA has been monitoring environmental impacts for many years directing studies at construction projects, remediation of spills and environmental impact assessments (Conlan *et al.*, 2004; Conlan *et al.*, 2010; Howington *et al.*, 1992; Lisle *et al.*, 2004; McFeters *et al.*, 1993). Likewise, Australia (Morris *et al.*, 2000), Brazil (Montone *et al.*, 2010), France (Delille and Delille, 2000), and the United Kingdom (Hughes and Thompson, 2004), have previously investigated contamination from wastewater outfalls. However, the majority of research to date has focused on small spatial scales, few time-series datasets have been collected (Kennicutt II *et al.*, 2010) and often a full suite of wastewater indicators have not been measured. In essence, there is the severe lack of monitoring data available for national Antarctic programmes (Gröndahl *et al.*, 2009; Hughes, 2010). This may be due to nations not submitting details of work through the Antarctic Treaty System's Electronic Information Exchange System, or more significantly, monitoring work is not being conducted at all.

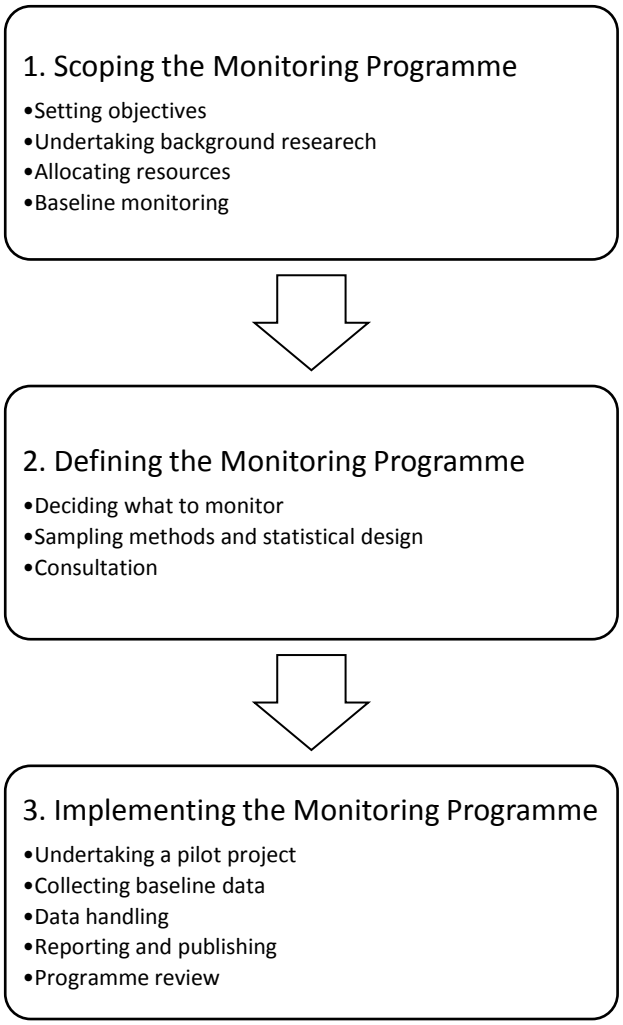


Figure 3. The Three-Step Approach to Environmental Monitoring developed by COMNAP.

1.2.3 Scott Base Sewage and Wastewater Monitoring Programme

New Zealand initiated monitoring environmental impacts at Scott Base in 1988. Preliminary research involved a survey of benthic invertebrates and mobile predators (Battershill, 1989). This was then followed by a comparison between the steep reef slope around Pram point and the flat reef of Cape Armitage, and also the effects of the wastewater discharge on local benthic communities (Battershill, 1992). This study identified that Pram Point was a suitable habitat for a diverse array of sedentary fauna at depths greater than 30m, yet in shallow areas, the sleep slope and loose substrate prevents large invertebrates being present. Additionally, the study found there to be little evidence that discharge from Scott Base negatively affected the marine benthic community (Battershill, 1992).

Anderson and Chague-Goff (1996) developed the monitoring studies further in 1994/95 by evaluating the effects of sewage and wastewater effluent discharges from Scott Base on the marine benthic environment. This study sampled sewage, sediment and wastewater for trace metals. Copper (Cu), Pb and Zn were found to be elevated in water and sediment near the effluent outfall. Benthic organisms were also

identified, with distribution and densities recorded across the reef slope. Anderson and Chague-Goff (1996) concluded that changes in foraminifera composition were due to environmental conditions, resulting from the discharge of effluent.

In the 1995/96 summer season, Antarctica New Zealand initiated annual monitoring of the effluent distribution in the marine environment at Pram Point. The programme measured faecal coliform, BOD and nutrient levels. Suspended solids were also measured in the first year, but discontinued as no clear trends were apparent. These surveys were conducted twice a year between the months of October to December. Sample sites were located in a grid, offshore from Scott Base and seawater samples were collected by drilling through the sea ice. Water samples were collected at a depth of one metre below the bottom of the sea ice (Redvers, 2000).

Antarctica New Zealand also started weekly monitoring in 1995/96 to include seawater at the reverse osmosis (RO) intake and raw effluent at the outfall. Seawater samples were analysed for faecal coliform bacteria and BOD, while faecal coliform and total suspended solids was measured in raw effluent (Redvers, 2000). To supplement this monitoring, Royds Consulting characterised the sewage and wastewater outputs to determine contaminant concentrations. Samples collected were analysed for suspended solids, BOD, faecal coliform, total nitrate (N) and total phosphorus (P). Royds Consulting concluded that the effluent discharged was variable in quality and quantity (Redvers, 2000).

Redvers (2000) followed with a more thorough monitoring regime over two successive summers to characterise the sewage plume and receiving waters. The suite of parameters used for this study was much broader than previous monitoring, and included faecal coliform, total N, total P, total organic carbon (TOC), BOD, Cu and Zn. The study also investigated the effect of current patterns and vertical stratification on sewage dispersal. Redvers (2000) concluded that the plume of dispersion was restricted to an area 150 - 175 m long-shore and 40 - 50 m offshore, with weak and variable tide currents not providing effective dispersion of the wastewater.

Since Redvers (2000), there has been no water quality survey of the marine receiving environment at Pram Point. However, Antarctic New Zealand conduct monthly monitoring of the water entering the RO plant, drinking water, and treated wastewater prior to discharge. Faecal coliforms are counted for RO intake seawater, drinking water and treated wastewater. Total suspended solids are measured in drinking water and treated wastewater, while BOD is only measured in treated wastewater.

1.2.4 Implications for Monitoring the Effects of Wastewater Discharge

Anthropogenic impacts on a regional or continental scale are rarely examined in Antarctica as the majority of impact assessments are made on small scales and in single disciplines (Tin *et al.*, 2008). Environmental monitoring in the future requires a greater scope of research to develop not only an overview of all human impacts, but gathering information on the effectiveness of environmental impact assessments (Tin *et al.*, 2008). This is achieved with well designed routine monitoring programmes that

improve understanding of cumulative effects, provide reports on the state of the environment and demonstrate compliances with numerous water quality standards. Monitoring will also provide baseline data to better determine if contamination is introduced by anthropogenic or natural means.

1.3 Anthropogenic Impacts on the Antarctic Environment

1.3.1 The Terrestrial Environment

Approximately 0.3% of the Antarctic surface is ice-free terrain (Fox and Cooper, 1994) with the dominant biota consisting primarily of mosses, lichens, microinvertebrates and microorganisms (Hughes, 2010). These organisms are subject to numerous environmental stressors such as temperature, desiccation, nutrient limitation, radiation and physical abrasion (Convey *et al.*, 2008). As a result, terrestrial organisms in Antarctica are susceptible to disturbance and slow to recover (Gröndahl *et al.*, 2009).

The majority of the 120 research stations that have been built since the IGY are on ice-free coastal ground to allow easy construction and access to shipping (Hughes, 2010). The scraping of loose surface material, enhanced snow and ice melt, and changes in the permafrost are the main modifications to the landscape surrounding stations (Bargagli, 2000), and often restricted to a few square kilometres around the stations (Kennicutt II *et al.*, 2010; Walton, 1998). For example, the first mass terrestrial disturbance at Scott Base was on 10 January 1957, when a D8 bulldozer provided by Admiral Dufek from McMurdo Station, levelled the site where Scott Base now stands (Harrowfield, 1997).

In an era when environmental management was less stringent, the most common contamination of surficial soils near research stations comes from abandoned waste dumps and fuel spills (Tin *et al.*, 2008). At McMurdo Station, the spatial patterns of trace metals followed the trends of hydrocarbons, suggesting an anthropogenic origin, as arsenic (As), cadmium (Cd), Cu, Pb and Zn exceeded the background concentrations (Kennicutt II *et al.*, 2010). Likewise, transportation and deposition rates of trace metals were investigated by Suttie and Wolff (1993) who found that on a local scale, emissions from a running generator travelled relatively short distances. Pb, Cu, Cd and Zn concentrations were greatest near the exhaust, yet they were not detected beyond 40m downwind of the generator. The British stations Halley 4 and 5 were also investigated and as expected, the scale of impact is much greater. The same trace metals as stated previously were detectable within a 10km range.

Terrestrial contamination is highly variable from season to season with melt water runoff contributing greatly to the degree of dispersion (Tin *et al.*, 2008). Melt run off is variable, and often reduced due to constantly cold temperatures limiting the amount of liquid water available (Kennicutt II *et al.*, 2010). Regardless of the rate of melting, the above studies indicate that the contaminants can persist in the environment for an extended period of time.

1.3.2 The Coastal Marine Environment

Despite the waters of Antarctica being described as the least affected on the globe, there should be no reason for complacency as marine ecosystems worldwide are continually facing anthropogenic pressure (Halpern *et al.*, 2008). Being described as pristine has little value when what is being compared to is significantly degraded (Aronson *et al.*, 2011). Whilst on a continental scale pollution in Antarctica may be insignificant, locally it can be considerable (Elliott, 2005). Pollution arises from poor waste management, carelessness or accidents (Walton, 1998) and anthropogenic impacts on marine ecosystems occur at both local and regional scales (Aronson *et al.*, 2011). At present, the main known pollutant sources at Antarctic research stations are sewage outfalls, abandoned dump sites, accidental oil spills, and exhaust emissions (Bargagli, 2008). Sheppard et al. (1997) also recognised that terrestrial melt water runoff was another potential non-point source of contamination.

Sewage contamination in Antarctica has been typically found to be localised, occurring within 300m around sewage outfalls (Howington *et al.*, 1992; Hughes and Thompson, 2004). However, contaminants have also been observed at distances of up to 1000m along the shoreline (Howington *et al.*, 1992). In general, the extent of sewage pollution is influenced by an array of factors, including the number of people on the base, the effectiveness of wastewater treatment, and the oceanographic process effecting dispersion and dilution (Howington *et al.*, 1992; Hughes and Blenkharn, 2003; Stark *et al.*, 2003a)

The long term impact of sewage and wastewater in Antarctica is mainly unknown, as it has not been studied in great detail (Tin *et al.*, 2008). The following section provides an overview of the known effects of pollution in the Antarctic marine environment.

1.4 Consequences of Pollution on the Marine Environment

The impacts of wastewater discharge on the marine environment range from aesthetic, to more severe physical effects on ecosystems. In seawater, nutrients can cause microbial blooms, reducing oxygen levels; suspended solids can bury sessile organisms and be detrimental to feeding activities; and trace metals can cause pathological anomalies or decreased diversity (Aronson *et al.*, 2011). In general, exposure of marine organisms to wastewater contaminants may result in a diseased or stressed state (Smith and Riddle, 2009) and cause broader ecological impacts to marine communities (Evans *et al.*, 2000; Negri *et al.*, 2006; Stark *et al.*, 2003b).

The effect of contamination on marine ecosystems in polar environments is variable. Due to the rare habitat and the slow recovery rates of Antarctic marine species, it is thought that evolutionary adaption has resulted in organisms being more sensitive to anthropogenic stressors than temperate or tropical marine organisms (Chapman and Riddle, 2003; King and Riddle, 2001). In addition, understanding the ecological effects of wastewater is made more complex due to variable mixing regimes of the discharge area. In general, impacts may be greater if discharge is into shallow water or an enclosed bay (Tin *et al.*, 2008), yet altered current patterns make measuring impacts difficult.

The first report of anthropogenic pollution effecting marine biota in Antarctica was undertaken by Lenihan *et al.* (1990), and subsequent studies showed that even comparatively low volumes of sewage could affect the near-shore marine environment by introducing toxic contaminants and solid wastes (Crockett and White, 2003; Stark *et al.*, 2003a). Lenihan and Oliver (1995) illustrated that over a 15 year period, the offshore benthos was altered as a result of contamination from the McMurdo Station's waste, not natural variation. It was observed in Winter Quarters Bay that hydrocarbons, trace metals and PCB contamination altered benthic communities resulting in low infaunal and epifaunal abundance. This finding was of significance as it was the first study to link pollution with altered communities, rather than of seasonal ice scouring.

It is well understood that impacted sites have lower species abundance and biodiversity, and an increase in resistant and opportunistic species (Pearce and Wilson, 2003; Stark *et al.*, 2003a). Initially Dayton and Robilliard (1971) hypothesised that enrichment from sewage outfalls would favour mobile deposit feeders such as sea urchins and sea stars. This would result in larval recruitment success of many species being reduced, in particular sponges that dominate the epibenthos. However, Stark et al. (2003b) found that sediments contaminated with hydrocarbons and trace metals resulted in greater abundances in polycheates and gastropods. In regards to the physiological effects of pollution, it is possible that the ability for organisms to detoxify or remove contaminants from their bodies is reduced. This is because most biological functions of benthic organisms including growth, reproduction and metabolism, are slow in polar regions (Barnes and Clarke, 1995). On the contrary, organisms may also accumulate contamination at a slower rate (Duquesne *et al.*, 2000).

1.4.1 Microbial Pollution

Currently, the likelihood of disease in Antarctic organisms as a consequence of anthropogenic activity is considered to be quite low (Grimaldi *et al.*, 2011). However, sewage microorganisms have the potential to cause disease in marine mammal and bird populations. A recent study investigated *E. coli* present in two seal pups (6% of the sampled population), and it has been suggested that toxic microbes may be ingested by marine mammals. What is not clear is how the pathogen was introduced to the pristine environment. It was hypothesised that the source could be waste from nearby fishing vessels, or contaminated food and sewage from land-based stations or ships (Hernandez *et al.*, 2007). Although there are few reports of pathogenic microorganisms in Antarctic birds and mammals, recent studies suggest that increasing pressure on areas of colonisation could increase vectors of disease in the future.

Due to the characteristics of the colonies being favourable for rapid spread of microorganisms, the probability of microorganisms causing harm is increased (Grimaldi *et al.*, 2011). Also, the breeding colonies of most seabird and some seal species are established on ice-free coastal areas, along with many Antarctic research stations which could increase the probability of disease even further.

Gardner et al. (1997) investigated the occurrence of an avian pathogen in populations of Adélie penguin chicks, *Pygoscelis adeliae*. Infectious bursal disease virus is a pathogen of domestic chickens, which affects the lymphoid organs, and is highly infectious by the fecal-oral route. Gardner et al. (1997) found that the pathogen is resistant to inactivation by heat, desiccation and chemical agents. If poultry products are not disposed of properly, or sewage is not adequately treated, the pathogen could become viral with relative ease. This is once again of concern, as those ice free areas that are inhabited by humans, are breeding grounds for many avian species such as penguins and skewers.

1.4.2 Trace Metal Pollution

Most studies of trace metals in Antarctic waters have either investigated the processes that control the temporal and spatial distribution of natural sources of trace metals (Martin *et al.*, 1990; Nolting and De Baar, 1994; Westerlund and Ohman, 1991), or bioproduction and the role of bioactive metals in plankton ecology (De Baar *et al.*, 1995; Fitzwater *et al.*, 2000; Sanchez - Hernandez, 2000). For example, phosphate and Cd relationships are well understood in marine environments, and Cd is a biologically important trace metal that exhibits linear relationships with phosphate (Hendry *et al.*, 2008).

Results indicate that the spatial and temporal distribution of trace metals in Antarctic coastal waters are broadly different from that in the open ocean regions (Grotti *et al.*, 2001). Therefore, when comparing trace metal data from different sites, caution must be given to the location and environmental conditions which are present during sampling and if natural or anthropogenic sources exist. Consideration must also be given to the methodology used in analysis, and caution exercised when comparing results from different studies (Bargagli, 2000).

Natural metals come from remote regions via atmospheric deposition, or from physical rock weathering. Martin et al. (1990) compared the coastal area of the Gerlache Strait to the offshore Drake Passage and found that concentrations of Fe, Mn, Co, Cu, Cd, and Zn were greater in coastal waters. Similarly, Scarponi et al. (1995) investigated Pb and Cd concentrations in Terra Nova Bay, and found that the inlet near the Italian station had Pb values (61 to 114 pM) that were two to three times greater than those samples collected at a greater distance from the station or open ocean. A high spread of data was observed for Cd and it was noted that this is due to the known interaction with biological systems, and sourced from land born atmospheric dust.

Following from these studies, the focus shifted toward the behaviour of trace metals and what processes effect concentrations. Frache et al. (2001) reported that trace metals in the water column are affected by particulate matter entrapped in pack ice and released during ice melt and phytoplankton activity. The study still focused on natural sources of metals and neglected local anthropogenic sources. However, this is still important as background levels must be understood before deducing the magnitude of anthropogenic inputs. This has been done for other environments in Antarctica, for example a review by Bargagli (2000) noted that background levels of Pb are approximately 2 ng/kg in snow.

Once background levels are understood, the next step to assess anthropogenic impact is investigating at what levels trace metals become toxic to marine organisms, and if the natural background levels are sufficient to support life or if introduced trace metals actually a benefit for organisms. Trace metals such as Mn, Fe, Cu and Zn are required for optimal growth of orgnanisms (Lannuzel *et al.*, 2011). Grotti *et al.* (2005) noted that elevated concentrations of Fe may be linked to increased bio-productivity, phytoplankton biomass and nutrient drawdown, yet the source of Fe and other bioactive elements such as Zn, Mn, Ni, Cu and Cd is not well known. Large-scale and long-term programs such as the Biological Investigations of Marine Antarctic Systems and Stocks (BIOMASS), which have been running since 1977, provide strong and critical information on how pollution actually affects marine organisms and the characteristics of pollution in Antarctica (Sanchez - Hernandez, 2000).

Trace metals in marine organisms are typically accumulated within soft tissues, and toxic concentrations are significantly above background environmental levels. For example, free Zn ions in solution are also highly toxic to marine invertebrates and fish, and results in mortality of the organism. The planktonic crustacean *Daphnia* has been found to be highly susceptible to 6 micromolar of Zn, resulting in 93% of the population being killed (Sanchez - Hernandez, 2000). However, it has been realised that bioaccumulation is unique within each species and concentrations varied greatly between groups. Despite this, top predators have been recognised as the most effective bio-accumulators of trace metals, and thus a useful indicator of toxic concentrations arising from excessive discharge (De Moreno *et al.*, 1997; Illuminati *et al.*, 2010).

Ecotoxicolgical studies of Antarctic fish, *Trematomus bernacchii*, sampled from Winter Quarters Bay and a relatively pristine site showed increased rates of pathological anomalies due to anthropogenic impact (Evans *et al.*, 2000). However, it was concluded that levels of Zn, Cu and Cd in the liver were similar between sites, and that Ni was elevated at Winter Quarters Bay, but not to a sufficient level to cause a toxic effect. Thus, heavy metals at the polluted Winter Quarters Bay may only have a limited impact, and other contaminants from the wastewater outfall could be more influential in the physical condition of local vertebrates. In a subsequent study on *T. bernacchii*, Illuminati *et al*. (2010) investigated Cd accumulation and found that when exposed to seawater containing 2.0 mg/L of Cd, concentrations in tissue increased approximately 3.5 fold after 7 days. Results from these studies indicate that toxic effects may arise from trace metals from sewage outfalls or leaching from terrestrial sources. Despite *T. Bernacchii* having a relatively limited home range, the current concentrations of trace metals is not enough to cause significant impact on fish health (Evans *et al.*, 2000).

Similarly, Duquesne et al. (2000) investigated the response of the common amphipod *Paramorea walkeri* to toxicity and bioaccumulation characteristics when exposed to heavy metals. The species is pertinent as a study species as it is likely to be among the first organisms to be affected by contamination due to its habitat being in close proximity to the shore and also because macrobenthic organisms have been observed to ingest and assimilate sewage products (Conlan *et al.*, 2004). The results indicate that that *P.*

walkeri is sensitive to the toxic effects of Cu and Cd, and coupled with other characteristics, could be an ideal species to use as a biological indicator of heavy metal contaminants. However, what is first needed is investigation into how Antarctic organisms will be affected due to chronic exposure to pollutants. There has been widespread concern for susceptibility of organisms that evolved due to the unique ecophysiological characteristics. In particular PCBs and chlorinated pesticides have been outlined as highly toxic as they are lipophilic contaminants that will affect the fat-rich tissues of top predators (Sanchez - Hernandez, 2000).

Elevated concentrations of metals in organisms do not necessarily mean anthropogenic influence. The geochemical cycles of trace metals in Antarctica, as well as the life strategies of organisms is not well known at present (Sanchez - Hernandez, 2000). However, at Ross Island it is understood that due to basaltic volcanic rock in sediment, natural concentrations of Ni, Fe, Cu and Cr are prevalent, and Pb and Cd are low (Kennicutt II *et al.*, 1995). From the data available, it appears that Antarctic pollution is negligible and coupled with simple food webs, bioaccumulation processes may be reduced (De Moreno *et al.*, 1997).

1.4.3 Nutrient / Organic Pollution

Elevated levels of nutrient from wastewater discharge can alter marine ecosystems, especially in environments such as Antarctica that are normally nutrient-limited (Ellis-Evans *et al.*, 1997). The input of nutrients from wastewater discharge can promote microbial production, and in turn affect the photic zone depth, reduce dissolved oxygen, and cause bioaccumulation of organic and inorganic compounds (Danulat *et al.*, 2002; Hughes and Blenkharn, 2003; Smith *et al.*, 1994). The overall result of increased levels of organic material is increased respiration by heterotrophic bacteria. Subsequently, oxygen in the water column is consumed which reduces the amount of dissolved oxygen available (Smith and Riddle, 2009).

1.5 Waste Management in Antarctica

Despite discharging untreated wastewater into the environment, some stations still fulfil the obligations under the Madrid Protocol. Although the discharge from these small stations is negligible, the cumulative impact on the environment may be significant (Gröndahl *et al.*, 2009). As technologies advance, many methods have become more economical (EPA, 1999), and there is now little excuse for not treating wastewater from research stations. For example, biological treatment is a cost effective method that has been found to give a 200-fold reduction of *E. coli* (Reinthaler *et al.*, 2003). Wastewater treatment in Antarctica should also include disinfection as it is considered to be the principal method of inactivating or destructing pathogenic organisms (EPA, 1999). Using Ultraviolet disinfection could also be an attractive method for use in Antarctica as it is not a chemical process, thus eliminating the need for transport of toxic or hazardous chemicals. It also requires less space than other methods, and operation is reasonably straight forward (EPA, 1999). However, the characteristics of the wastewater, the intensity of the

radiation, or the time the pathogenic organisms are exposed to radiation all affect the effectiveness of disinfection (EPA, 1999).

1.5.1 Liquid Waste Streams

Sewage waste and 'grey water' originate from base toilets, laundry facilities, accommodation and cooking areas. The wastewater contains human excrement and associated nutrients, microorganisms (including potential pathogens), organic matter, detergents, heavy metals, disinfection by-products and hydrocarbons (Howington *et al.*, 1992; Hughes and Blenkharn, 2003; Smith and Riddle, 2009). The array of pollutants arising from sewage and grey water can have a broad spectrum of ecological impacts at many different trophic levels from, bacteria to invertebrates (Aronson *et al.*, 2011; Tin *et al.*, 2008). For the purpose of this study, the term wastewater will describe both sewage (human waste products) and domestic grey water.

1.5.2 Wastewater Treatment in Antarctica

Few nations go above and beyond what is required of them under Environmental Protocol to reduce their footprint as much as possible (Connor, 2008; Tin *et al.*, 2008), with New Zealand leading the way in managing and protecting the Antarctic environment. In a survey conducted by Grondahl *et al.* (2009), 63% of the permanent stations have some form of treatment system, yet only 31% of the summer stations have treatment systems. What is discouraging about the results of this survey is that only 48% of the 71 stations who completed the questionnaire have any type of waste water treatment at all (Table 1). Grondahl *et al.* (2009) also included results on the different type of treatment systems used (Table 2), and reported that of the 63% of stations that have treatment, the most common type is biological. Maceration and secondary treatment is also moderately common with 10% of stations using one or both of these treatment processes.

There are several challenges to wastewater treatment in Antarctica, and for this reason treatment facilities are lacking at many research stations. These include logistical difficulties, the need for trained personal, the cost of running and maintaining the WWTP, and the environmental conditions that exist in Antarctica. In a recent review of sewage disposal in Antarctica, Smith and Riddle (2009) outline that low temperatures cause reduced efficiency of some WWTPs, and heated facilities are required at great expense. Other recorded issues include the need treatment plants to be operated in exclusion from living and working areas, insulated and/or heated transfer lines, and the large seasonal variation in station populations requiring alterations of treatment methods. Other problems reported by Grondahl *et al.* (2009) were at Scott Base, outlining that it is often difficult to maintain a high quality of effluent during periods of increased occupancy on base during the summer. It has also been recognised that despite the best efforts to reduce water use on base, the waste water treatment plant requires a greater volume of water in order to run efficiently (pers. comm. Leitch, 2011). Additionally, running the plant is a

significant cost, requiring US$240 000 per annum for biological treatment to serve approximately 100 people, and a further US$9500 per annum to maintain (Gröndahl *et al.*, 2009).

The goal of wastewater treatment is to remove physical, chemical and biological contaminants from the wastewater, in order for the final effluent to be discharged into the marine environment. Typically, the process of wastewater treatment includes primary, secondary and tertiary phases. The primary phases of maceration, screening, and settling removes solids such as waste, fats, oil and grease. The secondary phase includes aerobic biological treatment where dissolved biological matter is converted into a solid mass, and the biological solids are disposed of or reused. Tertiary treatment disinfects the waste either chemically or physically, by means such as ultraviolet irradiation (Gogate and Pandit, 2004).

Table 1. Percentage of Antarctic research stations that have wastewater treatment (Gröndahl *et al.*, 2009).

Type of Station	No. of Stations	Sewage Treatment (%)
Permanent Stations	41	63
Summer Stations	26	31
Field Stations	4	0
All Stations	71	48

Table 2. Occurrence of Wastewater treatment types at Antarctic research stations (Gröndahl *et al.*, 2009).

Type of Treatment	Permanent Stations	Summer Stations
Primary	1	-
Secondary	4	3
Tertiary	1	-
Primary & Secondary	1	-
Primary, Secondary & Tertiary	1	-
Biological Plant	8	1
Septic Tank	2	3
Membrane & Biological	1	-
Chemical & Biological	1	-
Membrane & UV Filter	1	-
Electric Field & Disinfection	1	-
Maceration	4	-
Sewage Treatment Plant	-	1
No Treatment	15	18
Total Number of Stations	***41***	***26***

1.5.3 Wastewater Treatment at Pram Point, Ross Island

McMurdo Station Waste Water Treatment Practices

McMurdo Station is the largest research station on the continent (Negri *et al.*, 2006) and Winter Quarters Bay (WQB) is widely recognised as the most polluted marine site on the continent, as it served as the McMurdo Station dumpsite until the mid-1980s (Aronson *et al.*, 2011; Tin *et al.*, 2008). McMurdo Station's wastewater comes from typical domestic sources, and also from the maintenance facilities and research laboratories (Crockett, 1997). In the early 1990's the Division of Polar Programs of the National Science Foundation responded to pressure from the media and conservation organisations to reduce the increasing environmental impact at McMurdo Station (McFeters *et al.*, 1993). This lead to a sewage remediation plant being installed in 2003, at a cost of $1.2 million and a running cost of $125000 per annum. Previously there had been no treatment (Conlan *et al.*, 2004; Gröndahl *et al.*, 2009).

The waste water treatment process at McMurdo (Figure 4) includes three main phases; preliminary treatment of maceration, secondary treatment of extended aeration and then UV light to disinfect prior to being discharged to Winter Quarters Bay (Raytheon Polar Services, 2007). The system is capable of treating 495,900 litres per day, with a 757,900 litres a day peak flow (Biletnikoff *et al.*, 2006).

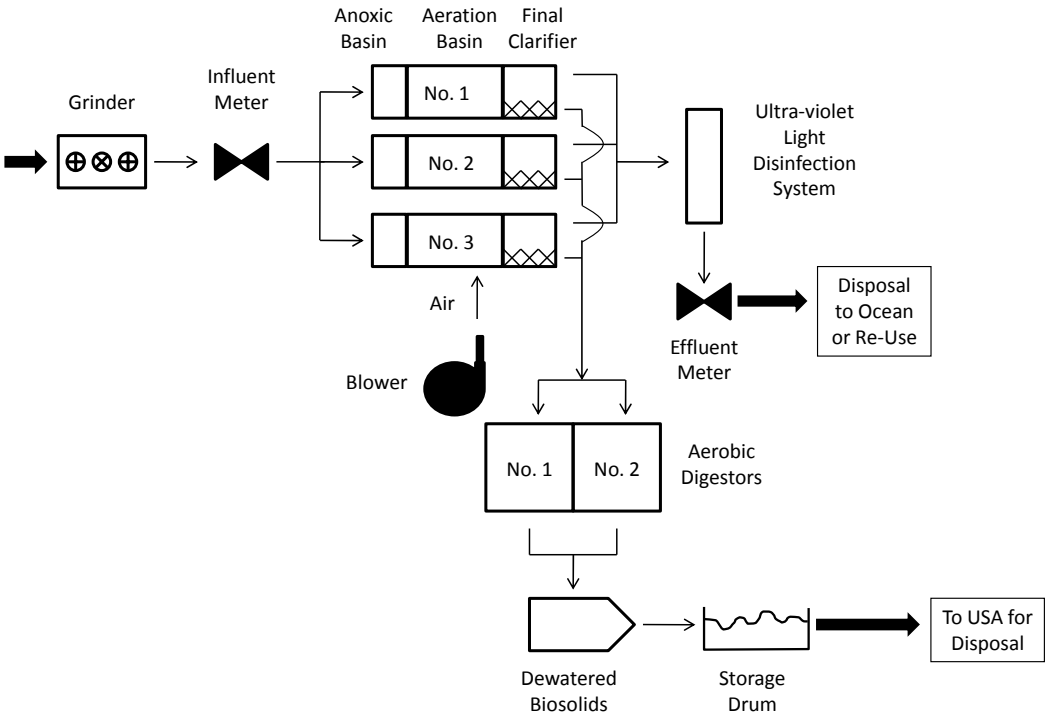


Figure 4. Schematic of the wastewater treatment plant, including the fixed media aeration treatment system, McMurdo Station, Antarctica. Figure redrawn from Biletnikoff et al. (2006).

Scott Base Waste Water Treatment Practices

Like many other bases in Antarctica, Scott Base discharged untreated sewage and wastewater into the local marine environment from its establishment in 1957 until 1998. Following the Madrid Protocol that came into force in 1998, effluent was macerated prior to discharge via a heated, insulated pipe. Prior to 14th April 1999, sewage and greywater were discharged from separate pipes, approximately 13m from the shore, where the liquid stream flowed overland, melting through the sea ice to enter the water column. Sewage discharge was intermittent, occurring when storage tanks became full and greywater was discharged directly from the kitchen (Antarctica New Zealand, 2001; Redvers, 2000). This method of disposal met the minimum requirement under Article 5.1 (b) of the Antarctica (Environmental Protection) Act, 1994.

In 2000, the outfall was moved to discharge through a tide crack approximately 5m offshore directly into the sea (Figure 5). In 2001, the untreated sewage and wastewater discharge met the minimum requirement of the Antarctica (Environmental Protection) Act 1994. However, to further support Antarctica New Zealand's environmental policy and to continue to reduce effects on the immediate marine environment, a new waste water treatment plant was installed in February 2002 and commissioned in October 2002. Antarctica New Zealand (2001) recognised that an increased quality treatment was required to reduce the effects on the environment, and avoid possible detrimental effects on aesthetic value, wilderness experience, scientific and biological significance. Antarctica New Zealand also elected to manage Scott Base under the same requirements for wastewater treatment in New Zealand. All sewage and greywater is now treated, and discharge is constant over a 24 hour period.

The new plant was designed to comply with World Health Organisation (WHO) minimum standards for safe swimming and best practice New Zealand standards (pers. comm. Leitch, 2011). The new plant is designed to be capable of treating a variable of flow ranges from 10 people in winter, to 120 in summer. The current wastewater treatment plant at Scott Base includes pre-clarifiers, fix bed aeration, and now treats the final phase of wastewater by Ozone instead of UV light. A schematic of the waste water treatment plant is presented in Figure 6.

Figure 5. The wastewater outfall after installation in early 2000 (Antarctica New Zealand, 2001). Since this time it has not moved and wastewater still discharges through the sea ice into the near shore environment.

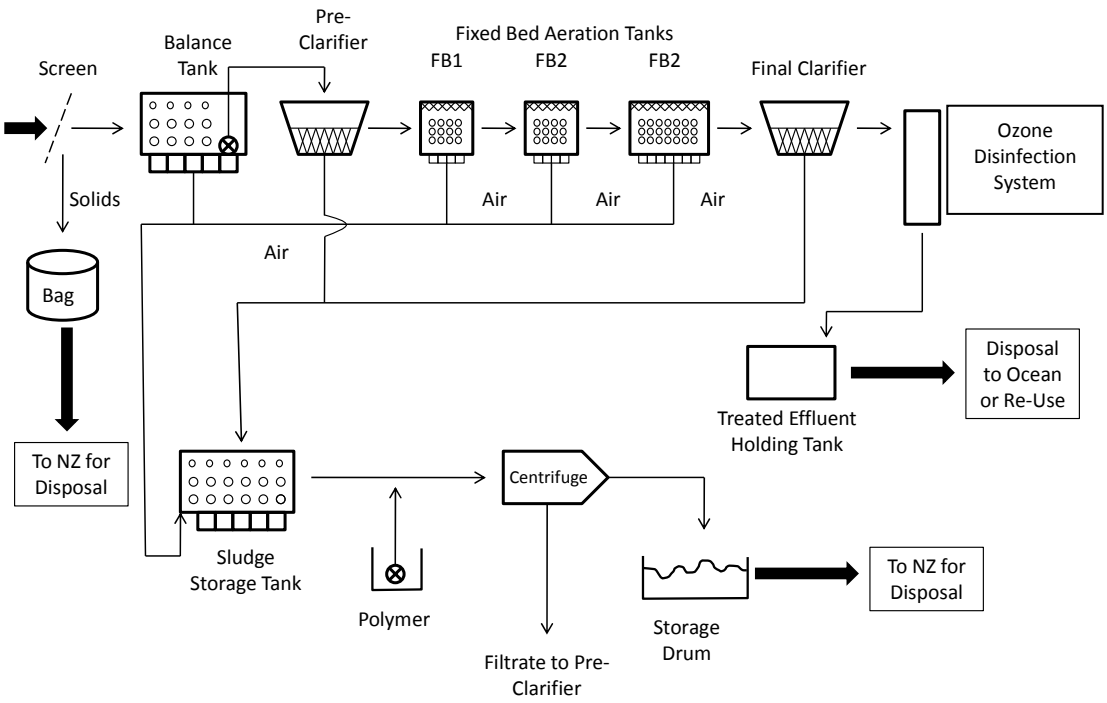


Figure 6. Schematic of the waste water treatment plant, including the fixed media aeration treatment system, Scott Base, Antarctica. Redrawn from Leitch (2011).

1.5.4 Sewage Discharge

On Ross Island, a significant amount of wastewater is discharged into the marine environment. Raw sewage from stations was either discharged into the nearshore marine environment, or buried in pits until the early 1960s (Aronson *et al.*, 2011). Prior to the installation of a wastewater treatment plant (WWTP) in 2003, McMurdo Base discharged between 58,000 and 300, 000 litres of macerated raw sewage, and 150, 000 litres of brine from the desalinisation plant each day during the months of September through to February (Lisle *et al.*, 2004). The installation of the WWTP has significantly improved the quality of the discharged wastewater (Table 3), with a pollutant removal rate of 90% or greater (Metcalf & Eddy Inc., 2008). Despite the increased water quality, there is limited distribution of the plume due to 3-5m of ice covering McMurdo Sound during the peak times of discharge (Lisle *et al.*, 2004). Scott Base by comparison discharged up to 17,000 litres of domestic wastewater daily prior to treatment (Waterhouse, 2001). At present, discharge is approximately 500 litres during the winter and 13,000 litres in summer into the local marine environment (pers. comm. Leitch, 2011).

Table 3. McMurdo Station Wastewater Characteristics. Adapted from Metcalf & Eddy Incorporated (2008).

Parameter	Untreated Wastewater Conc. (mg/L)	Treated Wastewater Conc. (mg/L)
TSS	264	14
BOD	398	3.1
Ammonia	22	1
Total Phosphorous	31	16.7

The level of sewage contamination surrounding outfalls depends on the biological and physical characteristics of the environment, the volume of discharge and the level of treatment (Tin *et al.*, 2008). While contamination has been found to be generally localised around sewage outfalls in polar environments to within a few hundred metres (Aronson *et al.*, 2011), studies show that even comparatively low volumes of sewage could affect the near-shore marine environment. Impacted sites generally have lower species richness, biodiversity and variability in benthic invertebrate communities compared with control sites (Stark *et al.*, 2003a). Genotoxic effects and pathological anomalies have also been reported in Antarctic fish exposed to sewage (Van Ngan *et al.*, 2007).

Previous monitoring programmes have used faecal bacterial indicators as a means of mapping the distribution and movement of sewage plumes from large Antarctic research stations (Howington *et al.*, 1992). However, with new technologies increasing the quality of the plume this may be difficult to achieve. Hughes (2004) also found that improved wastewater treatment at Rothera Research Station, Adelaide Island, Antarctic Peninsula, had reduced the microbiological pollution in the near-shore marine environment after one year of operation. This may be due to the altered discharge regime that is now in place, with slower but continual flow resulting in greater mixing of bacteria, or because conditions now

do not favour microbial reproduction. The waste water is not being stored in warm conditions that allow bacterial indicators to grow (Hughes and Blenkharn, 2003). Other indicators of wastewater that have proven to be useful are trace metals, such as copper and iron, pH and dissolved oxygen.

1.6 Rationale of Study

The aim of this study is to investigate the extent and characteristics of the wastewater discharge from Scott Base, and assess the efficiency of improved wastewater treatment operations in the last 10 years. Results from the current study will be compared to those of Redvers (2000) and to Antarctica New Zealand's historical data (Antarctica New Zealand, 2011) to assess whether the changes in treatment method have improved wastewater quality and its impact on the marine environment. Additionally, soil samples will be analysed to identify sources of trace metal contamination from the terrestrial environment, and drinking water quality at Scott Base tested for any trace of waste water entering the RO intake.

1.6.1 Study Objectives

The primary research objectives are:

I. To characterise the wastewater discharge from Scott Base and the water quality within the discharge plume beneath the sea ice.
II. To map the extent of the wastewater plume from Scott Base.
III. To determine if the extent or quality of the plume has changed using Redvers (2000) and Antarctica New Zealand's historical monitoring data as a baseline.
IV. To determine whether soils surrounding Scott Base are a potential source of trace metal discharge to the marine environment.
V. To characterise the reticulated freshwater at Scott Base to ensure no return of sewage discharge through the RO system.

The overall objective of this study is to provide information to Antarctica New Zealand on the potential impact of wastewater on the local marine environment. This study also fulfils Antarctica New Zealand's obligation under the Environmental Protocol to regularly monitor the environmental impacts of wastewater discharge. Information gathered can also be used to improve operating systems and as a tool to ultimately reduce the environmental footprint of Scott Base.

1.6.2 Research link with New Zealand Antarctic and Southern Ocean, Directions and Priorities 2010-2020

The document 'New Zealand Antarctic and Southern Ocean, Directions and Priorities' developed by the New Zealand Government outlines key directions and priorities for the New Zealand Antarctic and Southern Ocean science programme in the next ten years. The framework sets research goals within three broad 'Science areas' covering climate, terrestrial and coastal ecosystems, and the marine environment.

This research primarily investigates the environmental impact of Scott Base and links with the document's objectives listed for the inland and coastal ecosystem domain.

The context for these 'Science Areas', are set within the Antarctic Treaty System, including the Protocol on Environmental Protection (Protocol) and the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR). This research will address the protection of the Antarctic environment and associated ecosystems (Protocol) and proposes to provide a platform upon which more comprehensive studies of Antarctic marine ecosystems can be conducted in the future (CCAMLR), in particular along the Pram Point coastline.

The research presented in this thesis also links with the unifying themes outlined in the New Zealand and Southern Ocean Science Directions and Priorities by incorporating multidisciplinary research to further investigate the effects of Scott Base on the marine environment. The research domain that this study aligns with is Outcome 2: Inland & Coastal Ecosystems, whereby an improved understanding of inland and coastal ecosystems of the Ross Sea Region is sought. Under this outcome, research goals include an increased understanding of how the Antarctic environment may respond to climate change and other human impacts. This study will address the stated outcomes and goals by investigating the impact Scott Base has on the coastal marine environment to help understanding the response of Antarctic flora and fauna to anthropogenic effects.

This research investigates if current measures are adequately reducing the footprint of Scott Base and to what degree Antarctica New Zealand is meeting international obligations under the Protocol on Environmental Protection. It will provide recommendations for 'best-practice' to further reduce the footprint and promote world-leading environmental standards for the activities of Antarctica New Zealand.

1.7 Thesis Structure

Chapter two outlines the general study design, sample collection, and the analysis of both water and soil samples. Chapter three characterises the extent and quality of the wastewater plume from Scott Base. This is followed by results and discussion on the concentration and potential leaching of trace metals in soil in chapter four. Chapter five investigates if the reticulated water at Scott Base is affected by the wastewater outfall, complies with Drinking-Water Standards for New Zealand (Ministry of Health, 2008). Chapter six provides a synthesis of results, and outlines how this thesis can be used as a tool for improving environmental management in Antarctica.

2 Methods

2.1 General Study Design

Water and soil sampling was conducted off shore from Pram Point (Figure 7) from the 6th to 26th November 2010. Wastewater and seawater samples were collected for analyses of typical indicators of sewage contamination, including faecal coliforms (FC) /*Escherichia coli*, suspended solids, biological oxygen demand and trace metals. The population of Scott Base was approximately 60 - 80 people during this time.

Wastewater samples were collected to investigate concentrations of contamination before being introduced into the marine environment. Wastewater was collected prior to entering the wastewater treatment plant, and also after treatment. It should be noted that when comparing pre- and post-treatment data, the same 'parcel' of wastewater was not sampled. However, it is believed to be a fair assessment of how the wastewater treatment plant alters contaminants in wastewater.

Seawater samples were collected and analysed for typical indicators of wastewater. Field sites were selected on the basis of past marine monitoring (Redvers, 2000), and located offshore from Pram Point. The sampling sites were a maximum of 125m from the outfall, and included the RO intake.

Seawater samples were also collected at the RO intake, with freshwater samples of domestic (RO1) and drinking water (RO2) being collected on the Base. This was to understand if contaminants from the wastewater plume were reaching the RO intake, and if reverse osmosis was effectively treating water to an acceptable standard (Ministry of Health, 2008).

New terrestrial monitoring sites were also included, as indicated on Figure 7. Surface soils from around Scott Base were collected to investigate the potential for leaching of trace metals by melt water.

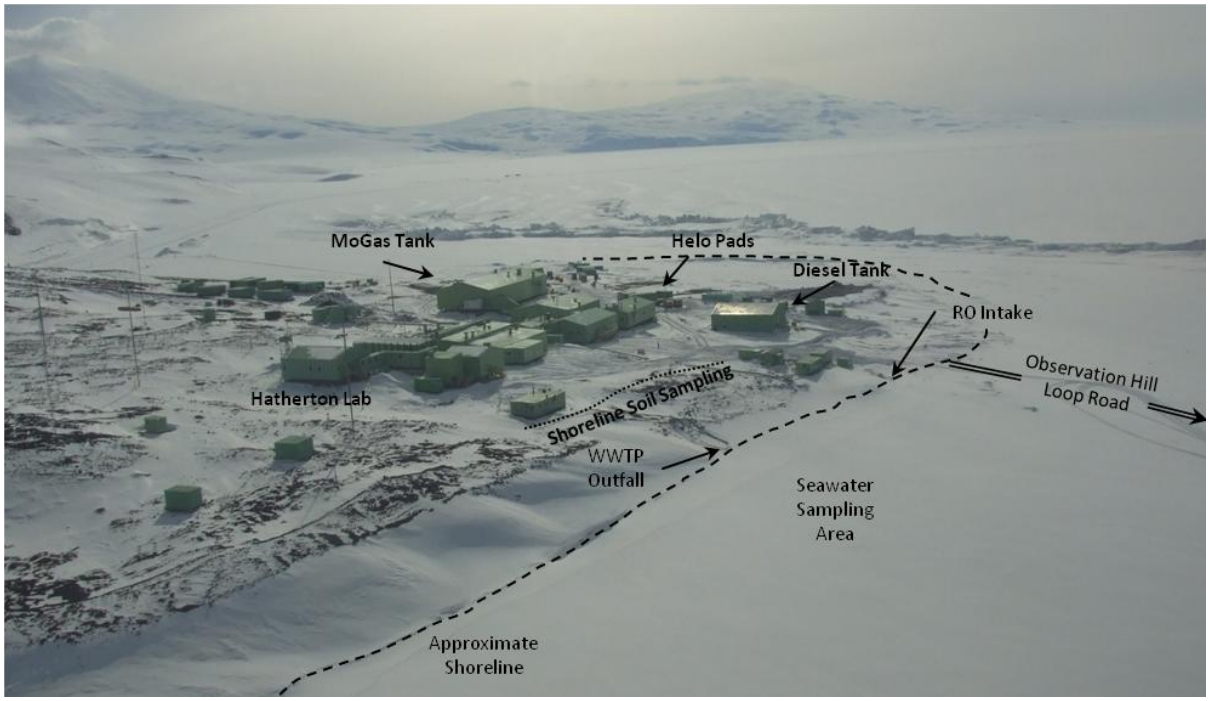


Figure 7. Aerial photograph of Scott Base, Antarctica (Rack, 2012). Locations on shore indicate where soil samples were collected, and seawater samples were collected offshore from the WWTP outfall.

2.2 Statistical Analysis

Statistical analysis was performed on all parameters, excluding trace metals for which there were insufficient data, to investigate the relationship between the distance from the outfall and depth on the concentration of selected parameters. Microsoft Excel, 2010 was used to perform a Pearson correlation.

2.3 Quality Assurance and Quality Control

Quality assurance (QA) and quality control (QC) ensures that work is congruent with the goals of the study, and that it is undertaken in accordance with the appropriate standards and protocols. This will ensure that reliable, accurate and comparable data is collected, a critical objective for any Antarctic monitoring programme.

A control site was located at a McMurdo Sound fishing hut, located in the McMurdo Sound. For each parameter, the QA/QC measures are included in the relevant methodology section. Overall, one duplicate sample was collected daily for each sample to assess variability.

2.4 Sample Collection

2.4.1 Site Locations

Wastewater samples were collected before and after treatment on two occasions (10th and 20th November 2010).

Marine sampling sites were chosen on the basis of past monitoring programmes (Anderson and Chague-Goff, 1996; Redvers, 2000). The sampling grid was modified from that used by Redvers (2000) in November 1999 (Figure 8). Sites 5, 7 and 19 were located in close proximity to the Observation Hill Loop Road and were excluded as clearing snow and drilling close to the road would have potentially caused parts of the road to melt out rendering it inaccessible. The location of the WWTP outfall or RO intake has not moved since Redvers (2000) study, yet seasonal changes to the sea ice has continually shifted the location of the pressure ridges resulting in the road position changing also.

Sites were located by using scaled maps which were previously surveyed into local benchmarks SBA and BMA (Redvers, 2000). All coordinates were in relation to the Scott Base survey grid (metres East and metres North) and are illustrated in Figure 8. GPS coordinates were also recorded, which would allow sites to be readily revisited for future site monitoring and are embedded in Figure 9. Seawater samples were also collected at three sites in Winter Quarters Bay for comparison with Scott Base (Figure 10).

Soil sampling sites were chosen on the basis of potential point sources of contamination. These sites were located near fuel storage or refuelling sites, or in areas that may affect the marine environment. Samples were collected along a transect to the south of Scott Base and parallel to the shore, as well as from the Mogas refuelling site, diesel tank refuelling site, Hatherton Laboratory and helicopter pad (Figure 11). As in previous studies (Kennicutt II *et al.*, 2010), sites were intentionally biased towards areas of known contamination, or potential contamination, and thus cannot be used to estimate the area of contaminated soil at Scott Base.

Drinking (RO2) water samples were collected in the dining area, from the water cooler, and RO1 samples were collected from the ablutions block.

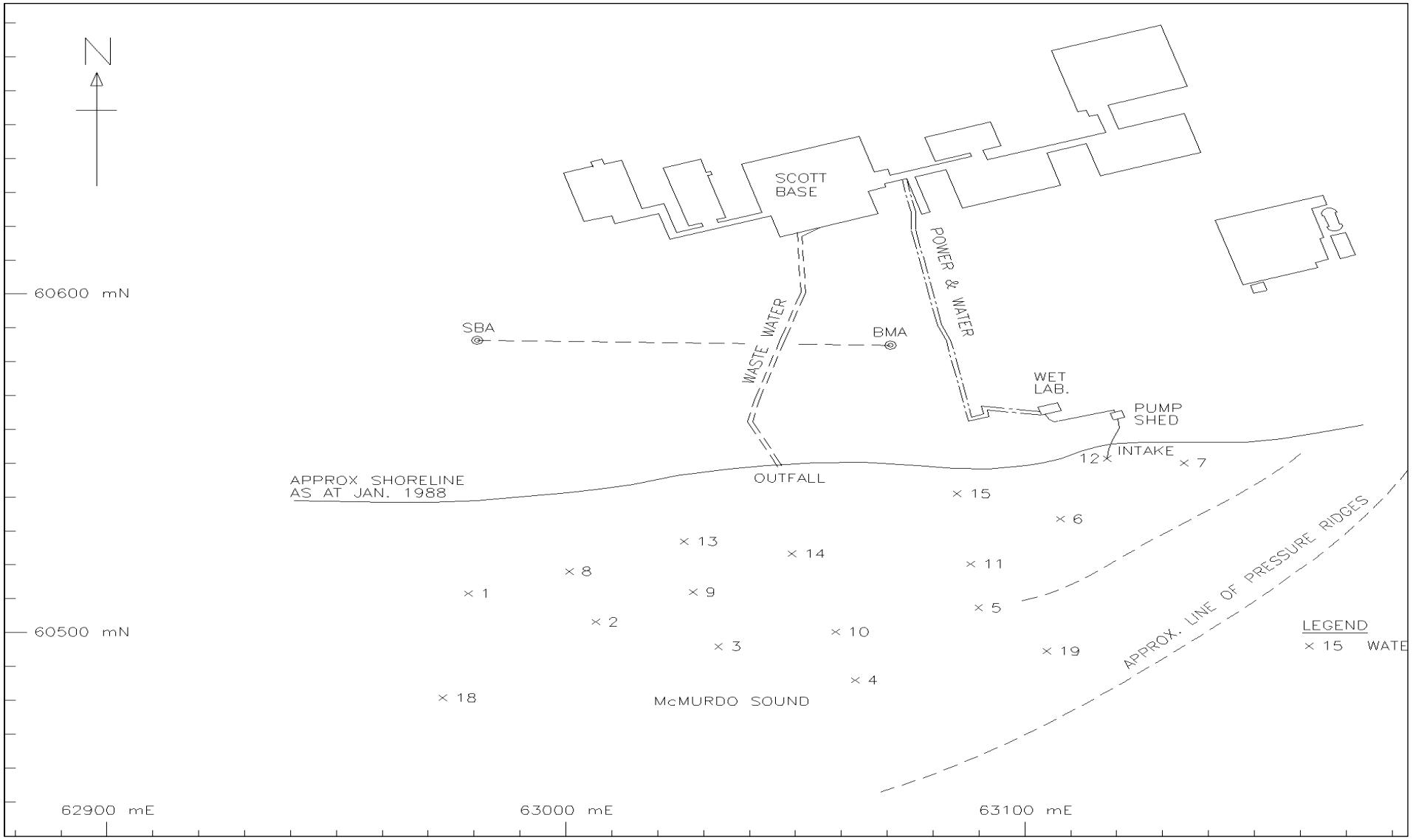


Figure 8. Seawater sampling sites used by Redvers (2000) in the 1999 summer season

Site	S	E
1	- 77 51.052	166 45.787
2	-77 51.054	166 45.880
3	-77 51.057	166 46.008
4	-77 51.053	166 46.047
5	-77 51.040	166 46.109
6	-77 51.025	166 46.126
8	-77 51.043	166 45.840
9	-77 51.048	166 45. 931
10	-77 51.046	166 46.007
11	-77 51.035	166 46.105
12	-77 51.008	166 46.142
13	-77 51.037	166 45.894
14	-77 51.031	166 45.963
15	-77 51.019	166 46.058
18	-77 51.085	166 45.829

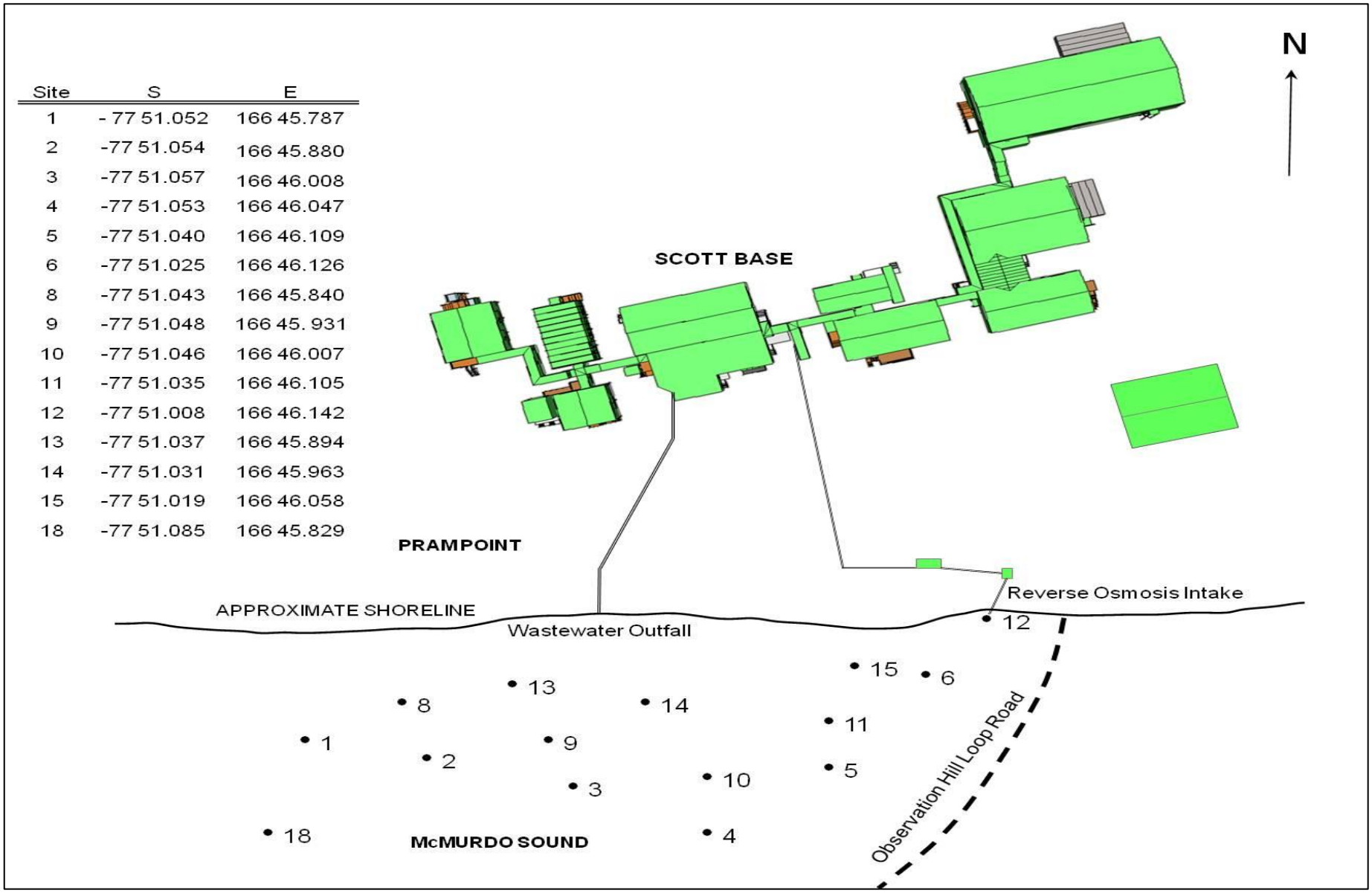


Figure 9. . Location of seawater sampling sites, Pram Point, Antarctica. Adapted from Antarctica New Zealand 3D model of Scott Base.

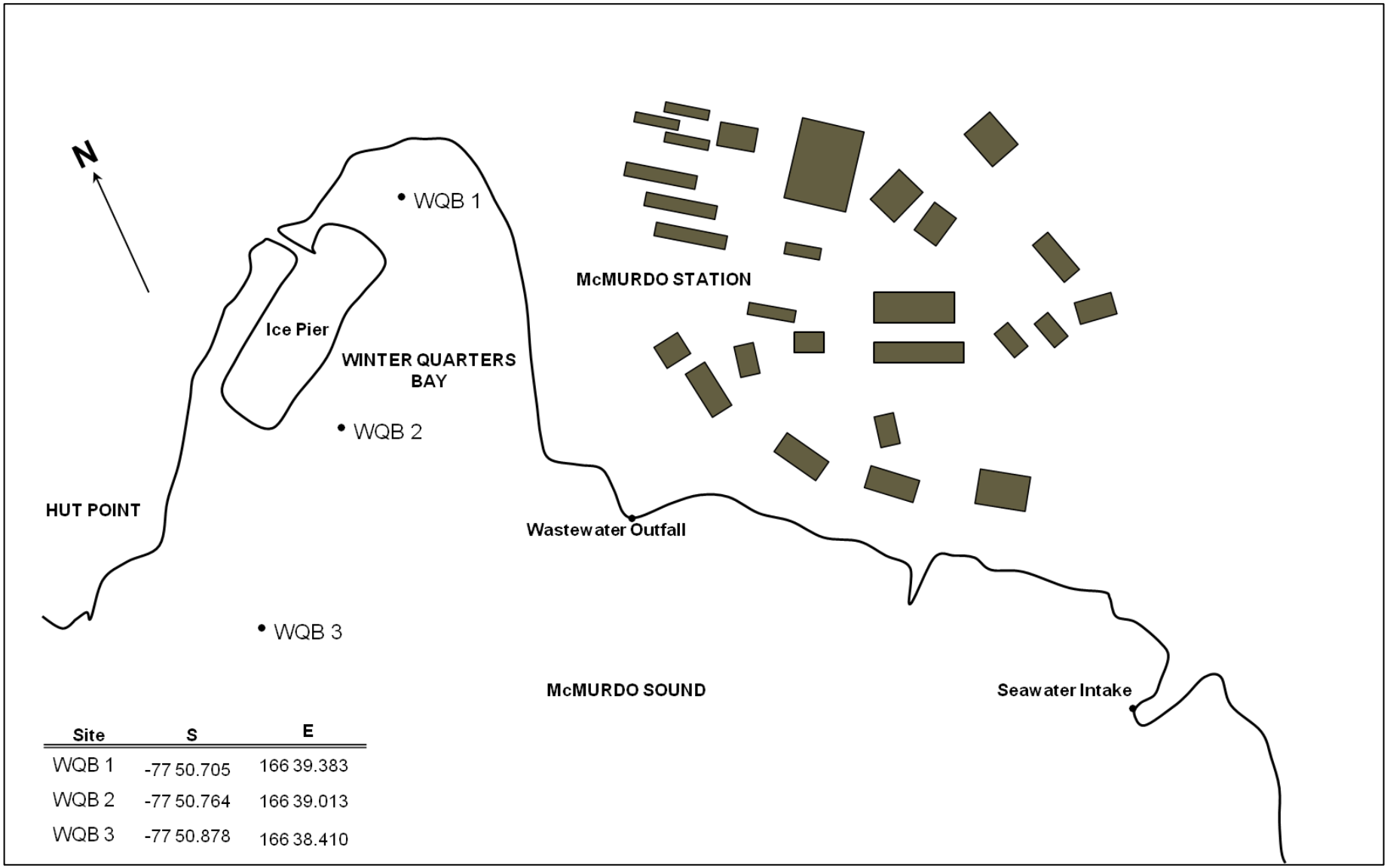
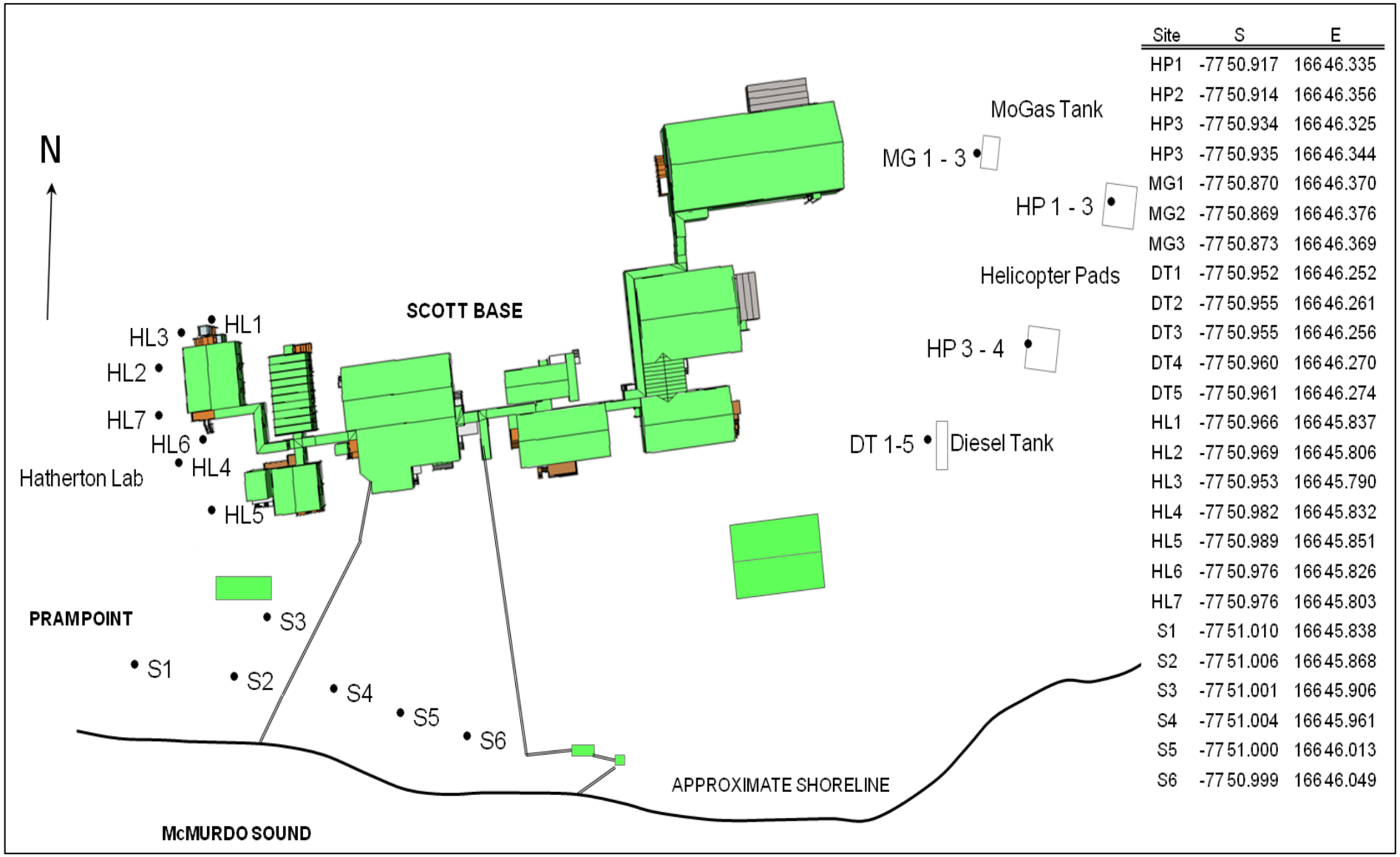


Figure 10. Seawater sampling sites at Winter Quarters Bay, McMurdo Sounds, Antarctica. Adapted from Railsback (1992)



Site	S	E
HP1	-77 50.917	166 46.335
HP2	-77 50.914	166 46.356
HP3	-77 50.934	166 46.325
HP3	-77 50.935	166 46.344
MG1	-77 50.870	166 46.370
MG2	-77 50.869	166 46.376
MG3	-77 50.873	166 46.369
DT1	-77 50.952	166 46.252
DT2	-77 50.955	166 46.261
DT3	-77 50.955	166 46.256
DT4	-77 50.960	166 46.270
DT5	-77 50.961	166 46.274
HL1	-77 50.966	166 45.837
HL2	-77 50.969	166 45.806
HL3	-77 50.953	166 45.790
HL4	-77 50.982	166 45.832
HL5	-77 50.989	166 45.851
HL6	-77 50.976	166 45.826
HL7	-77 50.976	166 45.803
S1	-77 51.010	166 45.838
S2	-77 51.006	166 45.868
S3	-77 51.001	166 45.906
S4	-77 51.004	166 45.961
S5	-77 51.000	166 46.013
S6	-77 50.999	166 46.049

Figure 11. Soil sampling sites in relation to Scott Base and surrounding features. Adapted from Antarctica New Zealand 3D model of Scott Base

2.4.2 Field Sampling

Marine water sampling was conducted at the same period of the tide each day (Figure 12), despite Redvers (2000) concluding that tidal conditions did not significantly alter concentrations of wastewater indicators. Sites were sampled starting from those furthest from the outfall and sampling towards the outfall in order to reduce the possibility of cross contamination between sites. Sites were cleared of snow (up to 2m deep) by Piston Bully and holes drilled through the sea ice, using a 25cm diameter auger connected to a 'Jiffy' drill (Figure 13). Samples were collected using a non-metallic Wildoco Teflon Kemmerer water sampler (Figure 14), at a range of depths from the top (1m beneath ice/water interface), middle and bottom of each profile.

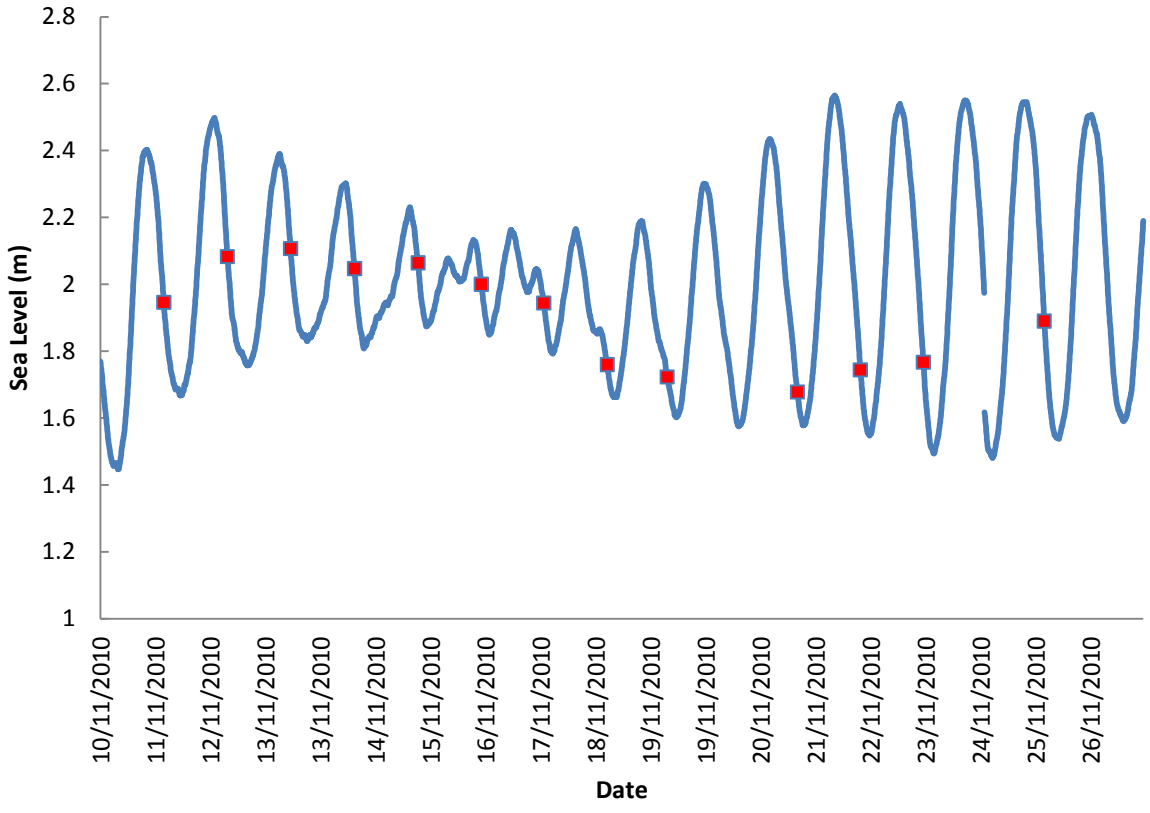


Figure 12. Sea level at Scott Base during the sampling period, with red marks indicating when sampling was conducted on the out-going tide.

Figure 13. Drilling through sea ice with Jiffy drill. The pits were cleared with the Piston Bully Snowcat.

Figure 14. Once the water sampler was retrieved, samples were collected and stored before analysis or transportation.

2.5 Water Analysis

Water samples were transported to laboratory facilities at Scott Base and refrigerated within two hours of being sampled. 50ml samples for trace metal analysis were stored in polyethylene centrifuge tubes and 50ml TOC samples were stored in brown glass bottles. All samples were transported to New Zealand in chilly bins and refrigerated at the University of Canterbury in MAF PC2 laboratories, or Hills Laboratories in Hamilton.

2.5.1 Faecal coliform/ *Escherichia coli*

Water samples were tested for sewage bacteria using 3M Petrifilm plates following the manufacturer's protocol (3M, 2008) at Scott Base, within one hour of collection. The plates were placed on a level surface, and 1ml of sample was delivered onto the centre of the film with the pipette perpendicular to the plate. The top of the film was then carefully rolled down over the sample to ensure even spread and avoiding entrapping air bubbles. The plates were incubated for 24 hours at 35^{o}C, then counted as per The Association of Official Agricultural Chemists (AOAC) International guidelines. Coliforms are gram-negative rods that produce acid and gas from lactose during metabolic fermentation. For this method, the acid produced coliform colonies growing on the Petrifilm plate causing the pH indicator to turn the gel colour a dark red or blue.

2.5.2 Trace Metals

Trace metals (^{66}Zn, ^{63}Cu, ^{55}Mn, ^{56}Fe, and ^{60}Ni) in marine water were determined by Inductively Coupled Plasma – Mass Spectrometry (ICP-MS). The system used was the Agilent 7500cx ICP-MS, equipped with an octopole collision cell (Agilent Technologies, USA). Measurements were carried out in collision gas mode with a helium flow of 5.5 ml/min and ^{103}Rh was added on-line as the internal standard.

The 50ml trace metal sample was stored in polyethylene centrifuge tubes and transported to New Zealand for analysis in University of Canterbury, MAF PC2 laboratories. 15ml of each sample was filtered then acidified to pH <1.5 for dissolved metals, and 15ml of sample was acidified for 24hours then filtered (as above) for total acid soluble metals. Filtration and acidification of all samples was through Satorius Stedium 0.45 µm minisart filters attached to Chirana Luer 20ml sterile hypodermic syringe. After filtration the samples could be removed from the MAF PC2 laboratory.

All samples were then diluted 21 times with 2% HNO_3 prior to analysis to reduce their high salinity. Two field blanks and one transport blank were also included in ICP-MS analysis, as well as spiked duplicates and triplicates to monitor ICP-MS performance. The method detection limits for trace metals were calculated by using taking the standard deviation of the ICP-MS blanks, and are presented in Table 4.

Table 4. ICP-MS spectrometry detection limits for trace metal analytes in µg/L

Trace Element	Cu	Fe	Mn	Ni	Zn
Detection Limit (µg/L)	1.2	17	0.3	0.4	12

QA/QC

Ambient levels of trace metals were accounted for as part of the QA/QC program. Transport blanks were carried to and from Antarctica and unopened, although poor packaging prior to deployment meant these samples were potentially compromised. Field blanks of milli-q water were used to assess for trace metal contamination by ambient environmental conditions. These were opened and exposed to the environment for approximately the same period of time as seawater samples, before being refrigerated and returned to New Zealand for analysis.

Analytical quality was carefully monitored, with duplicates, recoveries of spiked samples and analysis of certified reference material. Duplicates were within 2.5% of each other, except Pb which was within 15%.

Four samples were sent to Hill Laboratories for trace metal analysis to provide an inter-laboratory comparison of the ICP-MS analysis, and results are reported below in Table 5.

Table 5. Comparison ICP-MS results for the same samples, by the University of Canterbury (this study) and by Hills Laboratories in Hamilton

Site	Fe (µg/L)		Cu (µg/L)		Zn (µg/L)		Cd (µg/L)		Pb (µg/L)	
	UC	Hills	UC	Hills	UC	Hills	UC	Hills	UC	Hills
T.12.0	<16.84	<84	<1.16	<21	<11.73	<84	0.11	<4.2	<7.54	<21
T.13.0	18.07	<84	2.1	<21	34.92	105	0.36	<4.2	<7.54	<21
T.15.15	35.04	<84	26.49	46.2	27.18	<84	0.37	<4.2	<7.54	<21
Blank	<16.84	<84	4.76	<21	<11.73	<84	0.18	<4.2	<7.54	<21

Unfortunately, much of the trace metal data was below the detection limit for the Hills Laboratories ICP-MS. The UC detection limites were lower, but this low concentration data has been regarded as possibly unreliable in the interpretation due to the potential for polyatomic interference when analysing seawater.

2.5.3 Total Suspended Solids

Clean, GFC filter papers were dried for 24 hours at a temperature of 35^{o}C, and their weight recorded. 1000ml of seawater and 250ml of effluent sample was then filtered under vacuum through the filter to collect all suspended solids of > 0.45µm size. The filter paper was then dried at 103˚C for a further 24 hours prior to re-weighing and the suspended sediment content was calculated as the weight difference.

The initial 600ml of filtrate was disposed of, and the remaining 400ml of filtrate was kept for nutrient analysis by HACH colorimeter (refer to section 2.5.4).

2.5.4 Nutrients

At Scott Base, a portable HACH DR/890 colorimeter was used to measure concentrations of nitrate - nitrogen (NO_3^-- N) and phosphate (PO_4^{3-}). HACH Method 8039 was used for NO_3^-- N and method 8048 for PO_4^{3-}.

A cell was filled with 10ml of filtered sample, and the appropriate Reagent Powder Pillow was added. The sample was shaken vigorously for one minute, and then left to stand for five minutes for NO_3^-- N reactions and two minutes for PO_4^{3-} reactions. The cell was filled with an untreated sample to zero the meter, and at the completion of the reaction time, the sample absorbance was read. Sample cells were rinsed with milli-q water immediately after measurement to remove all cadmium particles and the waste was returned to New Zealand to be autoclaved as per MAF PC2 requirements, and disposed of appropriately.

The manufactures estimated detection limit for NO_3^-- N was 0.8 mg/L, and 0.05 mg/L for PO_4^{3-} .

QA/QC

Nutrient checks were conducted every second day using premade standards. Milli-q samples were also analysed to check the blank analysis. Duplicates from one site were measured each day of sampling to monitor the performance of the portable colorimeter. The variation between duplicates of NO_3^-- N ranged from 0 mg/L to 0.2 mg/L and for PO_4^{3-} ranged from 0.04 mg/L to 0.13 mg/L. Nutrient standards were also sent to Hill Laboratories for validation (Table 6), which indicated that the HACH method may overestimate PO_4 at low concentrations near to the detection limit..

Table 6. Inter-laboratory comparison of nutrient standard and blanks.

Nutrient	Standard		Blank 1		Blank 2	
	Hills	UC	Hills	UC	Hills	UC
NO_3^-- N	0.94	0.8	<0.002	<0.8	<0.002	<0.8
PO_4^{3-}	1.04	1.39	<0.02	0.3	<0.02	0.13

2.5.5 Biochemical Oxygen Demand

Biochemical oxygen demand (BOD) is a chemical bioassay method that measures the total amount of oxygen consumed by bacteria while they are consuming the organic matter in the sample, as well as chemical oxidation processes. The BOD was measured using current Antarctica New Zealand protocol and OxiTop methods. The water sample was shaken vigorously for one minute to ensure complete mixing and saturation of oxygen, then 365ml of sample was measured into the OxiTop bottles and five drops of nitrification inhibitor was added to the sample. An autoclaved magnetic stirrer was added, and a quiver containing two sodium hydroxide pellets was inserted into the neck of the bottle. OxiTop units

were then attached to the bottles and placed into an incubator, and after ensuring the stirrer was working, left for five days. The OxiTop units automatically record the level of BOD over the selected time period, and in this case, reading on the fifth day for the BOD value.

2.5.6 Total Organic Carbon

Water samples of approximately 250ml were stored in dark brown glass total organic carbon (TOC) bottles, to be transported to New Zealand and analysed at Hill Laboratories Ltd. in Hamilton. Total carbon was analysed by catalytic oxidation and infrared detection. Total inorganic carbon was analysed by acidification and purging. TOC was calculated by subtracting total inorganic carbon from total carbon as per APHA 5310 b guidelines. The default detection limit for this analysis was 0.5 mg/L TOC.

2.5.7 pH/Dissolved Oxygen/Conductivity

The pH, dissolved oxygen (DO) and conductivity samples were collected in clean plastic bottles and measured immediately upon bringing a sample to the surface, using a HACH HQ40d Portable Meter.

QA/QC

The accuracy of the pH and conductivity calibrations were checked every three days by returning the electrode to one of the calibration solutions and measuring the parameter concerned. The electrode was then rinsed thoroughly prior to measuring subsequent samples.

Duplicates at one site were measured for each day of sampling to monitor the performance of the portable meter. The variation between duplicates for pH ranged from 0 - 0.01, DO from 0 - 0.6 mg/L and conductivity from 0 - 0.5 mS/cm.

2.5.8 Turbidity

Turbidity was measured in seawater immediately upon bringing the sample to the surface, or straight from the WWTP using a HACH DR/890 portable spectrometer (method 8237). Turbidity is a qualitative characteristic which is determined by suspended solids obstructing the transmittance of light through a water sample. Usually it is measured in nephelometric turbidity units (NTU) but in this case it has been measured as formazin attenuation units (FAU). However, the two units are comparable and there is a strong correlation between NTU and FAU (pers. comm.Webster-Brown, 2012). Approximate NTU concentrations are also given in the results.

2.6 Soil Analysis

Soil samples of approximately 200g were collected using a plastic trowel and air-dried at Scott Base. Soil samples were transported to the University of Canterbury for further air-drying and analysis in a MAF PC2 laboratory.

2.6.1 Soil Metal Concentrations

Metal concentrations in terrestrial soil were determined using USEPA Method 200.8. Approximately half of the dried soil sample was sieved through a plastic sieve with a mesh diameter of 2mm. For each sample, 1 gram of soil was weighed out into acid washed polycarbonate centrifuge tubes. The sample was then digested in 10ml of 1+4 HCL and 4ml of 1+1 HNO_3 and heated under reflux at 90°C for 30 minutes. The sample was left to cool and then made up to 20 ml with Milli-q water. The sample was diluted 10 times with 2% HNO_3 for ICP-MS analysis as described in Section 2.5.2. Detection limits outlined in Table 4 were adopted, with additional detection limits for As (0.56 µg/L), Cd (0.05 µg/L), Cr (1 µg/L) and Pb (6 µg/L).

2.6.2 Readily Leachable Metal Concentrations

The soil samples were leached with water using an adapted version of the US EPA's Method 1312 'Synthetic Precipitation Leaching Procedure' (US EPA, 1994). For each sample, 1 gram of soil was weighed out into acid washed polycarbonate centrifuge tubes and 20 mL of milli-q water was dispensed. The polycarbonate tubes were mixed in an end-over-end mixer for 18 hours at 40 rpm. Samples were centrifuged for 10 minutes at 5500 rpm to effectively settle the sediment out and the decant was filtered through Minisart <0.45 µm syringe filters and diluted five times with 2% HNO_3 prior to being analysed by ICP-MS as described in Section 2.5.2.

QA/QC

Matrix spike recoveries and certified reference material (CRM) recoveries were calculated for soil analysis. A standard reference material 2702 (National Institute of Standards and Technology, 2004) was used, and the % recovery calculated (Table 7).

Table 7. Certified reference material, trace metal concentrations, as specified (CRM), and as analysed in this study (UC) ± standard deviation (SD) with the recovery percentage.

Trace Metal	CRM	UC ± SD	% Recovery
Cr	352	243 ± 1	69
Mn	1757	1467 ± 5	83
Cu	117.7	91.2 ± 1.2	77.5
Zn	485.3	367.4 ± 3.3	75.7
As	45.3	37.1 ± 0.1	81.9
Cd	0.817	0.784 ± 0.013	91.54
Pb	132.8	113.9 ± 2.6	85.8

2.7 Errors in Analysis

Measurements of physical quantities are subject three general types of errors, and this study has attempted to account for such error. Random error is caused by fluctuations in environmental variables that are generally uncontrollable. Systematic error includes instrumental, methodological, or personal mistakes and can be identified and eliminated after inspection of the experimental methods, cross calibration of instruments, or examination of techniques. Lastly, gross errors are purely carelessness on behalf of the person conducting analysis, or equipment failure.

This study has adopted an error tolerance between 5-10% and will reject any value greater than 10%. This error tolerance is particularly important for interpreting trace metal results, as the probability of error is greater.

2.8 Difficulties with Working in the Antarctic Environment

Due to the remote and harsh conditions in Antarctica, monitoring programmes are often compromised (Kennicutt II *et al.*, 2010) by factors including the extreme cold and high winds. Trace metal analysis is especially difficult, because like snow, seawater contains low trace element concentrations and it is very difficult to avoid contamination during sampling, transportation and analysis (Bargagli, 2006).

Steps were taken to ensure water samples were not compromised during collection, transportation and possible contamination continually assessed, as outlined above.

Equipment failure in Antarctica is also an expected burden to researchers, and occurred at various times during this study. Fragile parts on the teflon water sampler did not survive the freezing conditions, and the critical bolts which hold the sampler together failed on one occasion. Similarly, the multi-probe for *in situ* temperature, pH, conductivity and DO was not used after the first sampling run, as frozen water around the probes prohibited them from working properly. For this reason, temperature data was unable to be recorded.

3 Characterisation of Wastewater Plume from Scott Base

3.1 Introduction

This chapter introduces the factors that influence the dispersion of effluent plumes, in particular in polar environments. The data collected in November 2010 is then presented, interpreted and compared with the results obtained by Redvers (2000). This comparison will determine if the improvements made to the wastewater treatment system at Scott Base have resulted in improved sea water quality in the local marine environment offshore from Pram Point.

3.1.1 Sewage Dispersion in Polar Marine Aquatic Environments

The physical behaviour of sewage dispersion from ocean outfalls is well understood in many environments. In temperate environments, initial dilution is caused by entrainment as the effluent rises up the water column, and dilution continues by natural oceanic processes (Wood *et al.*, 1993). However, many other factors can influence dispersion in polar environments. Within the coastal marine environment of the Southern Ocean, the dispersion and persistence of some contaminants can also be influenced by sea ice cover, low seawater temperatures (-1.8°C), solar radiation (Hughes, 2005), atmospheric dust loads and nutrient availability (Grotti *et al.*, 2001).

3.1.2 Previous Investigations of the Wastewater Plume from Scott Base

Initial investigations of the dispersion and fate of sewage and wastewater components from Scott Base indicated a localized plume of effluent contamination surrounding the outfall. From 1995 to 1997, Antarctica New Zealand conducted sea water monitoring offshore from Pram Point. Redvers (2000) continued monitoring in 1998 and 1999, with the first thorough investigation of marine water quality and wastewater dispersion offshore from Pram Point. Results from these studies indicate that the extent and characteristics of the wastewater plume have varied in space and time. Antarctica New Zealand's monitoring data showed that the plume extended up to a maximum of 100m offshore and up to 300m west of the outfall. Following the outfall being extended seaward, Redvers (2000) showed that the plume extended only 25-35m offshore, and 100-150 long shore. Redvers (2000) also concluded that the wastewater plume was buoyant due to the higher temperature of the discharge compared to the receiving environment.

As this study will compare results with Redvers'(2000) study, it is useful to understand the characteristics of the effluent discharge, marine water quality and the dispersion offshore from Scott Base in 1998 and 1999. Redvers (2000) found over the two successive summers that the highest concentrations of wastewater parameters in the marine environment were at sites closest to the outfall, and in most cases at the top of the water column. For reference, Redvers' (2000) maximum concentrations of all water quality parameters offshore from Scott Base are presented in Table 8. It should be noted that sites 14 and 15 are not exactly in the same position for both the 1998 and 1999 field season, but have not moved greatly.

Table 8. Maximum concentrations of wastewater parameters in marine water collected offshore from Scott Base by Redvers (2000).

Wastewater Parameter		Year	Depth (m)	Site	Maximum Concentration
Faecal coliform		1998	1	15	3550 cfu/100ml
		1999	1	14	2900 cfu/100ml
Trace Metals	Cu	1998	1	15	3.2 µg/L
	Zn	1998	1	15	4.8 µg/L
Nutrients	Total N	1998	1	15	2.2 mg/L
		1999	1	15	10.9 mg/L
	Total P	1998	1	15	0.52 mg/L
		1999	1	14	0.83 mg/L
Biochemical Oxygen Demand		1998	6	15	7.2 mg/L
		1999	1	14	6.1 mg/L
Total Organic Carbon		1998	1	15	2.6 mg/L
		1999	1	14	2.2 mg/L

3.2 Results of this Study

This section presents results of the extent and quality of the wastewater plume discharged from Scott Base in 2010. Samples were analysed for typical effluent indicators, including bacteria, nitrate-nitrogen, phosphate, pH, dissolved oxygen, trace metals and conductivity. To measure the extent of the wastewater plume, data analysis examined at what sites and depths concentrations were greatest, and if this correlated with the distance of the site from the outfall. Stratification of the water quality parameters in the water column was investigated to determine if the plume was buoyant. The results indicate an area in which the plume exists, yet the results from within the water column are often highly variable.

Sewage samples were also collected before and after treatment to investigate wastewater quality upon discharge, and it should be noted that pre- and post-treatment samples may not necessarily be the same 'parcel' of wastewater.

3.2.1 Faecal coliform / *Escherichia coli*

Marine water samples were collected from all sites offshore from Scott Base, as well as at the RO intake and at the WWTP (pre- and post-treatment). In the WWTP samples, bacteria counts were high with samples of 1:1 dilution too numerous to count (TNTC). Dilutions of 1:10 and 1:100 reduced the counts to a manageable level in the post-treatment samples, but not in the pre-treatment (Table 9).

Table 9. Faecal coliform and *E. coli* per 100ml pre- and post-treatment at Scott Base.

Dilution	Pre-Treatment / 100ml		Post-Treatment / 100ml	
	E. Coli	**FC**	***E. Coli***	**FC**
1:1	TNTC	TNTC	TNTC	TNTC
1:10	TNTC	TNTC	112000	145000
1:100	TNTC	TNTC	190000	410000

Samples collected at all sites and all depths in the marine environment offshore from Scott Base did not contain any faecal coliform (FC) or *Escherichia coli*. Similarly, seawater samples collected at the RO intake pump house did not contain FC or *E. coli.* Samples collected at Winter Quarters Bay and at the control site also did not contained FC or *E. Coli.*

3.2.2 Trace Metals

To assess concentrations of trace metals in the marine environment, an understanding of concentrations in wastewater before and after treatment must first be investigated (Table 10 and Figure 15). On the first sampling occasion, total concentrations of Cu, Fe, Ni, and Mn were greater in post-treatment samples, with only total concentrations of Zn being (very slightly) reduced after treatment. For all elements, dissolved concentrations were greater in samples collected after wastewater treatment. By comparison, in the second sampling occasion both total and dissolved concentrations of Cu, Ni and Zn were greater in samples collected after treatment. Fe and Mn had total and dissolved concentrations which were slightly reduced in post-treatment samples.

Table 10. Concentrations of total and dissolved trace metals collected in pre- and post-treatment wastewater samples from Scott Base.

Date Sampled	Sample Site		Trace Element (µg/L)				
			Cu	Fe	Ni	Mn	Zn
10/11/2010	Pre-Treatment	Total	6.0	255.1	10.3	43.8	62.9
		Dissolved	5.7	233.5	9.1	39.2	53.1
	Post-Treatment	Total	8.5	257.3	11.0	59.2	60.8
		Dissolved	6.0	258.1	10.5	56.0	59.7
20/11/2010	Pre-Treatment	Total	6.0	203.3	8.9	37.9	50.5
		Dissolved	4.5	196.9	7.8	36.1	45.4
	Post-Treatment	Total	6.8	186.0	9.3	33.2	62.5
		Dissolved	7.2	190.7	8.8	30.3	50.2

Marine water was analysed for trace metals at representative sites offshore from Scott Base. The results of trace metal analysis on the seawater samples are presented in Figure 16. Those sites that have no data presented in Figure 14 are below the detection limit. All trace metal data is presented in Appendix 1.

The concentrations of Cu, Mn, and Zn were greatest at site 13 to the west of the outfall, and site 15 to the east of the outfall. These sites were those closest to the shore (see map, Figure 8). Fe and Ni were detected at most sites within the plume, with Ni also at sites 13 and 15. Fe, Mn and Cu were also detectable or slightly enriched at site 14.

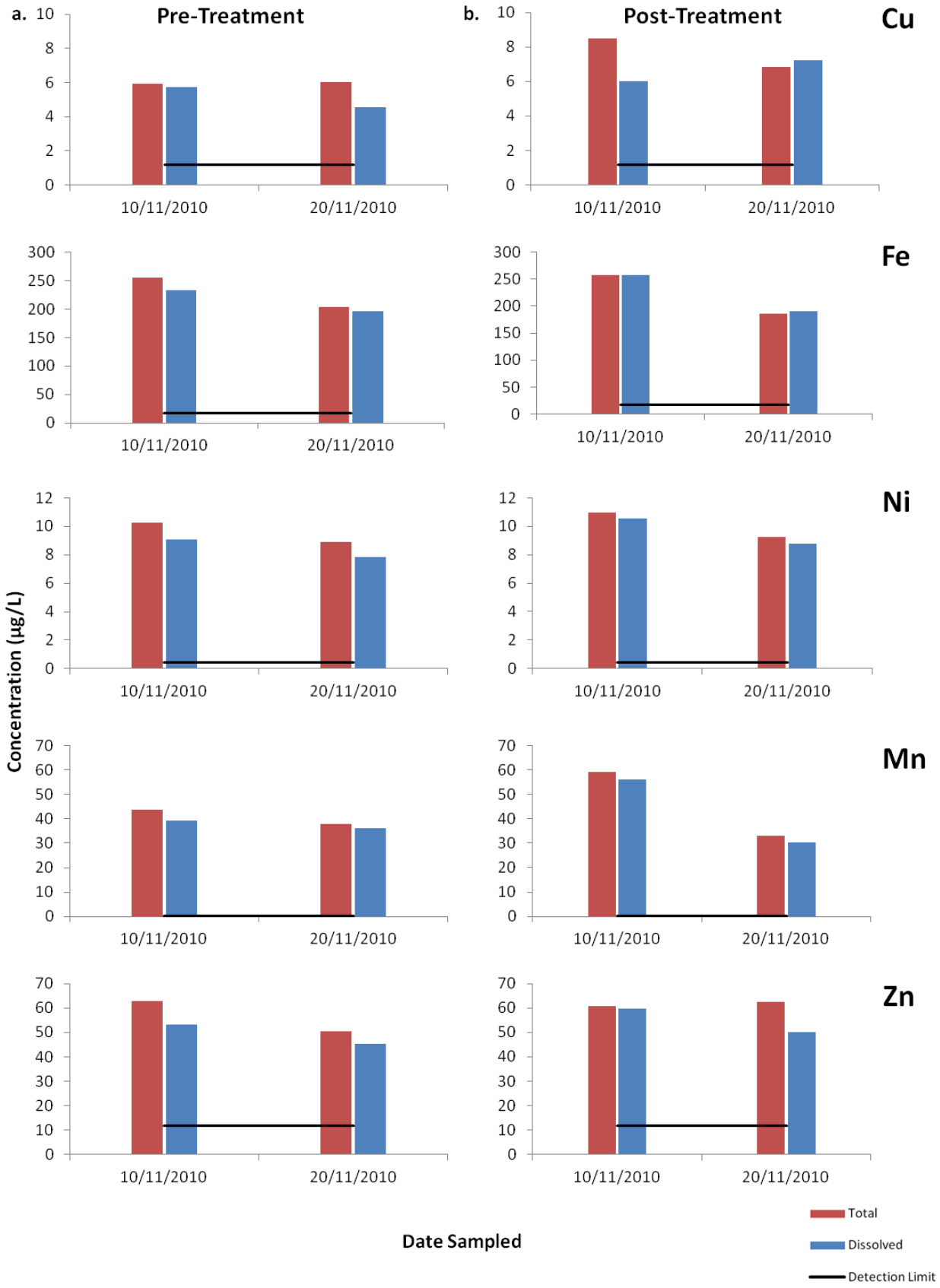


Figure 15. Concentrations of trace metals in wastewater samples collected in prior (a) and after (b) treatment.

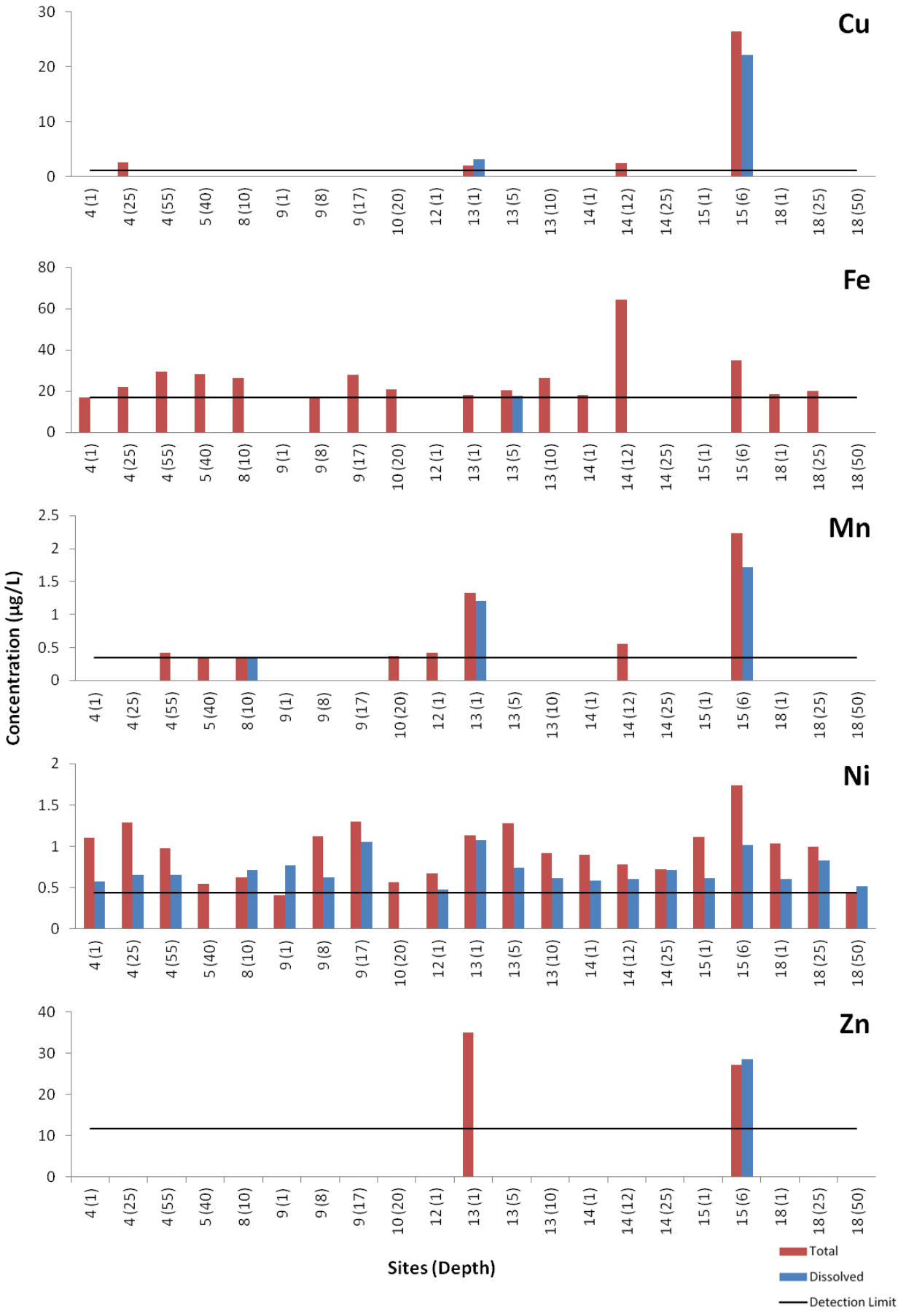


Figure 16. Concentrations of trace metals in seawater samples collected at Pram Point, Antarctica. Sites are indicated on the x-axis, with the depth indicated in brackets. For example, '4(1)' indicates site 4, at a depth of 1m.

3.2.3 Nutrients

Nitrate - Nitrogen

Wastewater samples were collected on the 10/11/2010 and 20/11/2010, before and after treatment. On both occasions, the NO_3^-- N concentrations were greater than 5.5 mg/L (Table 11), the upper limit for the analytical method used.

Table 11. NO_3^-- N concentrations for pre- and post-treatment samples

Sample Date	Pre-Treatment (mg/L)	Post-Treatment (mg/L)
10-11-2010	> 5.5	> 5.5
20-11-2010	> 5.5	> 5.5

In the marine environment, samples from sites 9 and 14 had NO_3^-- N concentrations greater than the estimated detection limit of 0.8 mg/L, as shown in Figure 17. These samples were taken at the top of the water column at site 14 (1.1 mg/L), and at site 9 in the middle and bottom of the water column (1 mg/L and 1.1 mg/L respectively). These are sites directly offshore from the outfall. The only other site to record a detectable NO_3^-- N concentration (0.8 mg/L) was at site 12 which is directly adjacent to the RO intake. This is consistent with the NO_3^-- N value of 0.8 mg/L measured at the RO intake pump house on the same day.

At the control site, all samples collected recorded concentrations below the estimated detection limit of 0.8 mg/L. The majority of samples collected at Winter Quarters Bay were below the detection limit, with only one sample from site 1 and a depth of 2m below the sea ice measuring 0.9 mg/L.

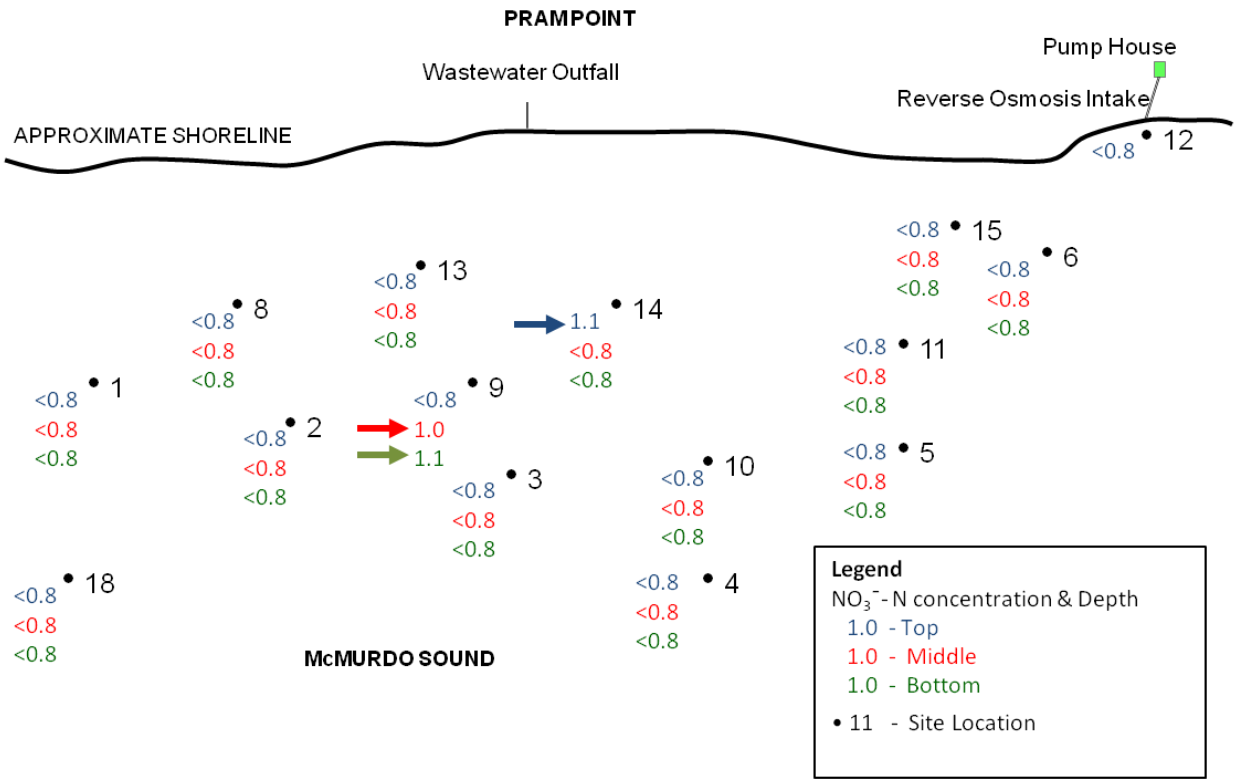


Figure 17. NO_3^-- N concentrations at each depth with site locations in relation to WWTP outfall and RO intake. Coloured arrows indicate the only detectable concentrations of NO_3^-- N.

Phosphate

PO_4^{3-} concentrations were also measured in wastewater pre- and post- treatment. For both sample occasions, the concentrations exceeded the maximum limits of the method (2.75 mg/L) (Table 12).

Table 12. PO_4^{3-} concentrations for pre- and post-treatment samples on two occasions.

Sample Date	Pre-Treatment (mg/L)	Post-Treatment (mg/L)
10-11-2010	> 2.75	> 2.75
20-11-2010	> 2.75	> 2.75

In the marine environment, the minimum and maximum concentration PO_4^{3-} recorded offshore from Pram Point was 0.28 mg/L and 0.8 mg/L respectively, with an average of 0.45 mg/L. Data analysis highlighted which sites recorded the greatest PO_4^{3-} concentrations, and at what depth. There was no clear pattern of dispersion, as the greatest PO_4^{3-} concentrations at each depth occurred at different sites (Figure 18). However, if the average of all depths at each site is calculated, a clearer pattern of dispersion is presented (Figure 19).

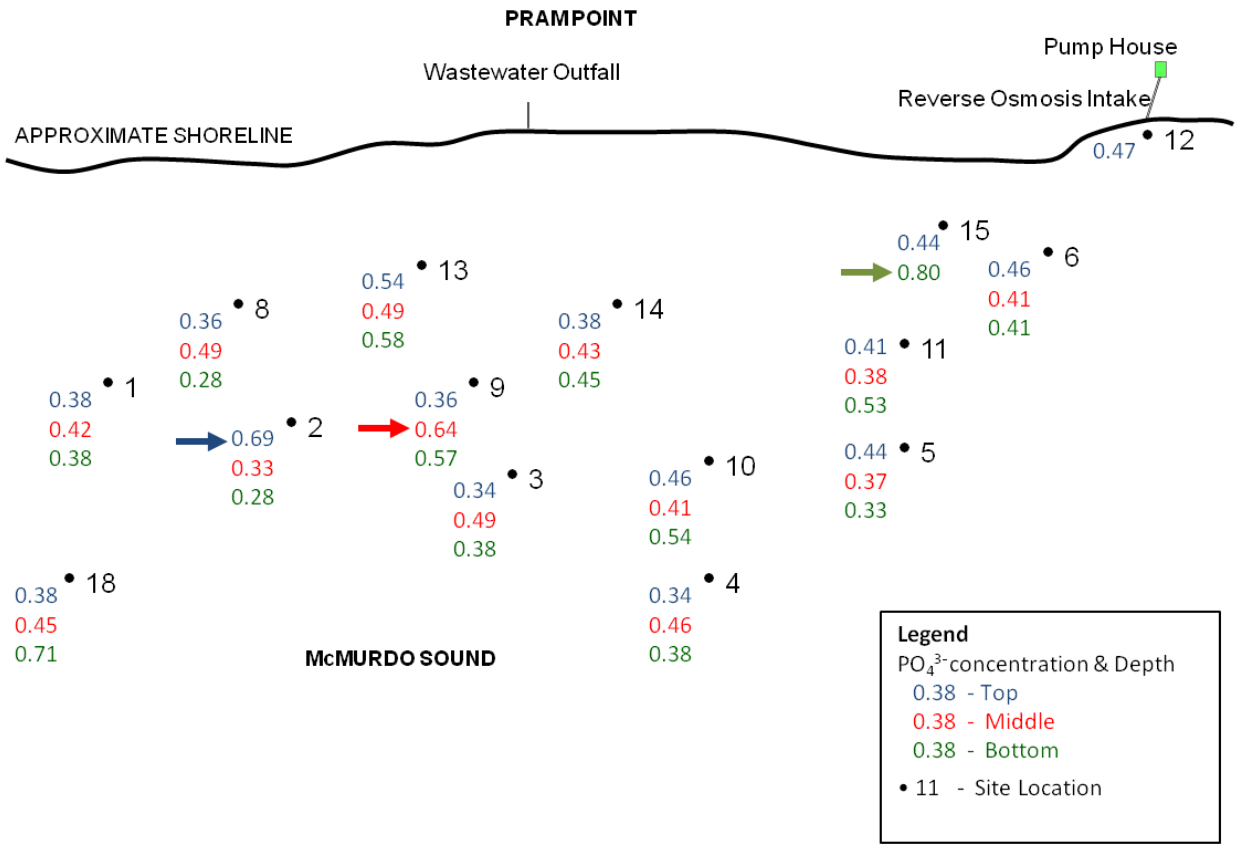


Figure 18. PO_4^{3-} concentrations at each depth with site locations in relation to WWTP outfall and RO intake. Coloured arrows indicate the greatest concentrations of PO_4^{3-} for each depth.

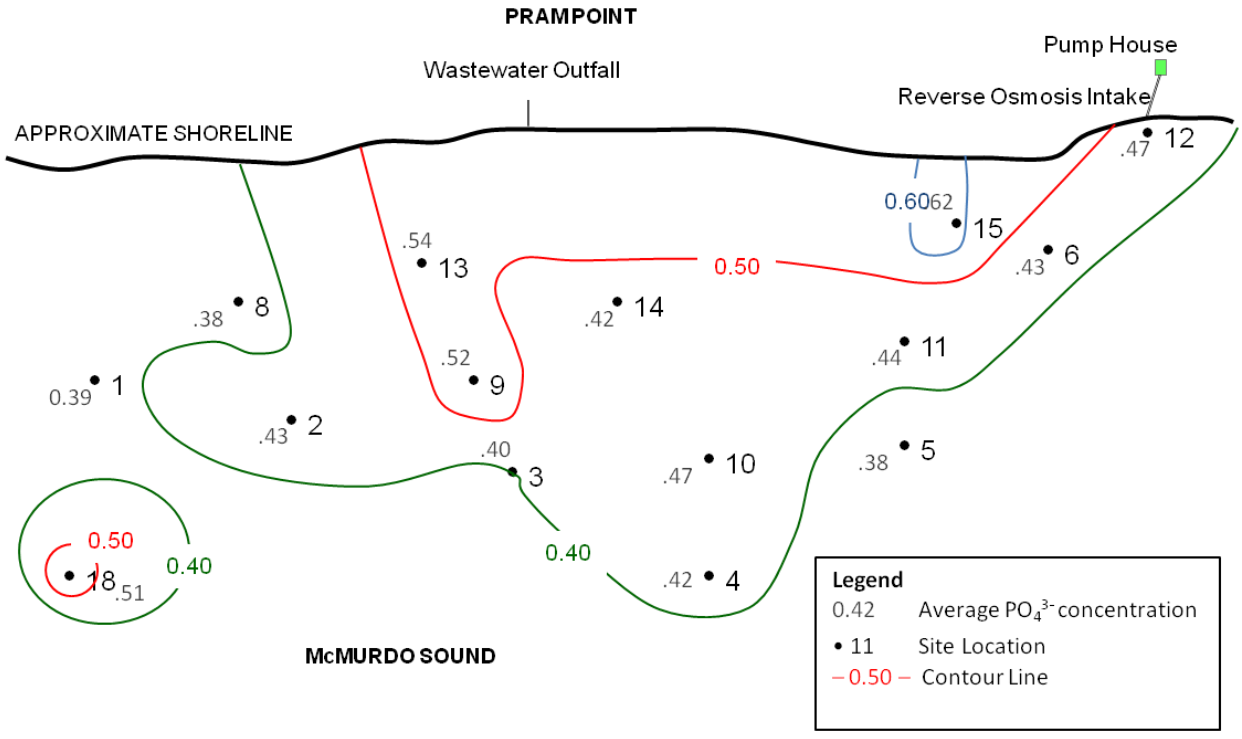


Figure 19. Average PO_4^{3-} concentrations over all depths, for each site with contour lines to indicate PO_4^{3-} distribution. Note that the contour lines are only an estimate, and that the concentrations in between the sites are not known.

To investigate the extent of the plume more rigorously, analysis investigated the correlation between PO_4^{3-} concentration and the distance of the site from the outfall. Pearsons correlation was performed on all sites at each depth, and also for an average of all depths at a site. There was no significant correlation between concentrations at each depth within the water column, and the distance from the outfall. Likewise, there was no significant correlation between an average concentration from all depths and the distance from the outfall.

Stratification of PO_4^{3-} in the water column was investigated by plotting depth against PO_4^{3-} concentration. At all sites stratification was occurring, and it was common for sites to be stratified such that the greatest concentrations were found at the bottom of the water column (Figure 20). Only four sites had the greatest concentrations at the top of the water column.

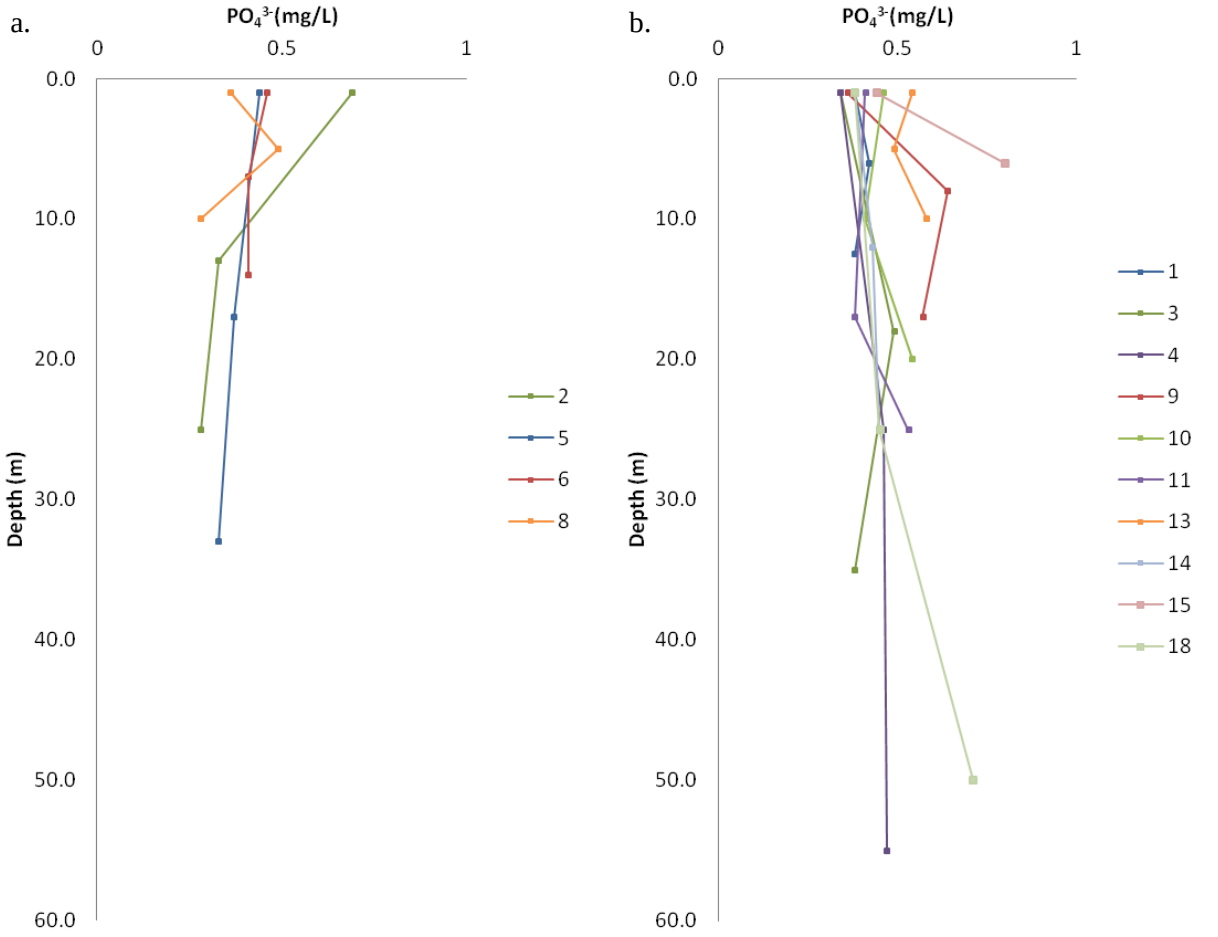


Figure 20. Depth profiles of PO_4^{3-} at sites that have decreasing concentrations with depth (a) and sites that have increasing concentrations with depth (b).

Seawater samples were also collected from Winter Quarters Bay (Table 13) and had a higher average concentration of PO_4^{3-} than those collected near Scott Base of 0.57 mg/L. The minimum and maximum concentrations were 0.34 mg/L and 0.95 mg/L respectively.

PO_4^{3-} concentrations collected from the control site (Table 14) were similar to those recorded near Scott Base, and on average similar to those from Winter Quarters Bay at 0.52 mg/L.

Table 13. PO_4^{3-} concentrations recorded at Winter Quarters Bay, McMurdo Sound.

Site	Depth (m)	PO_4^{3-} (mg/L)
1	1	0.46
	2	0.84
2	1	0.95
	9	0.35
	18	0.60
3	1	0.58
	15	0.44
	30	0.34

Table 14. PO_4^{3-} concentrations recorded at Control Site, McMurdo Sound.

Depth (m)	PO_4^{3-} (mg/L)
5	0.43
10	0.62
15	0.52

3.2.4 Dissolved Oxygen

The DO concentration of the wastewater pre-treatment was measured twice. The concentrations were 0.09 and 0.15 mg/L on the 10/11/2010 and 20/11/2010 respectively. Post-treatment, DO was 20.61 mg/L on the 10/11/2010 and 20.89 mg/L on the 20/11/2010.

In the marine environment, DO concentrations ranged from 10.05 mg/L at site 12 to 13.03 mg/L at site 18 (Figure 21), and averaged 12.04 mg/L across all sites. Investigating dispersion once again looked at each depth individually for all sites. Results indicate for the top, middle and bottom depths, there is no clear pattern of dispersion, as there was no statistically significant correlation with DO concentration and the distance from the outfall.

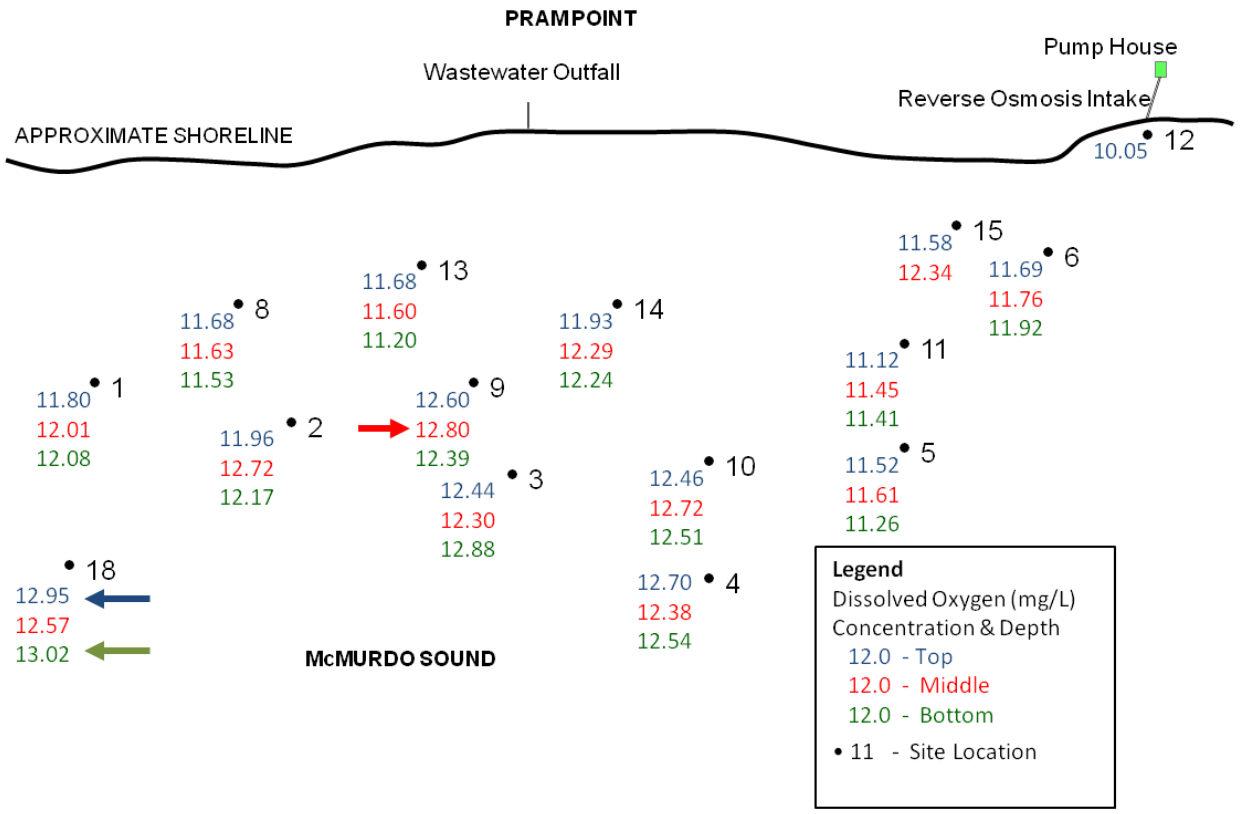


Figure 21. Dissolved oxygen concentrations at each depth with site locations in relation to WWTP outfall and RO intake. Coloured arrows indicate the greatest concentrations of DO for each depth.

Stratification of dissolved oxygen was measured at all sites offshore from Scott Base. The minority of sites had higher concentrations of DO at the top of the water column, with three sites clustered to the south-west of the outfall (8, 9 and 13) and two outlying (4 and 5) showing a decrease of DO with depth. All other sites increased in DO with depth (Figure 22).

Samples from Winter Quarters Bay (Table 15) had higher average concentrations of DO with 12.83. Both Scott Base and Winter Quarters Bay both had concentrations above the control site average of 11.73 mg/L.

Table 15. Dissolved oxygen concentrations of seawater collected at Winter Quarters Bay McMurdo Sound.

Site	Depth (m)	DO (mg/L)
1	1	12.69
	2	13.12
2	1	12.54
	9	12.96
	18	13.2
3	1	12.88
	15	12.84
	30	12.56

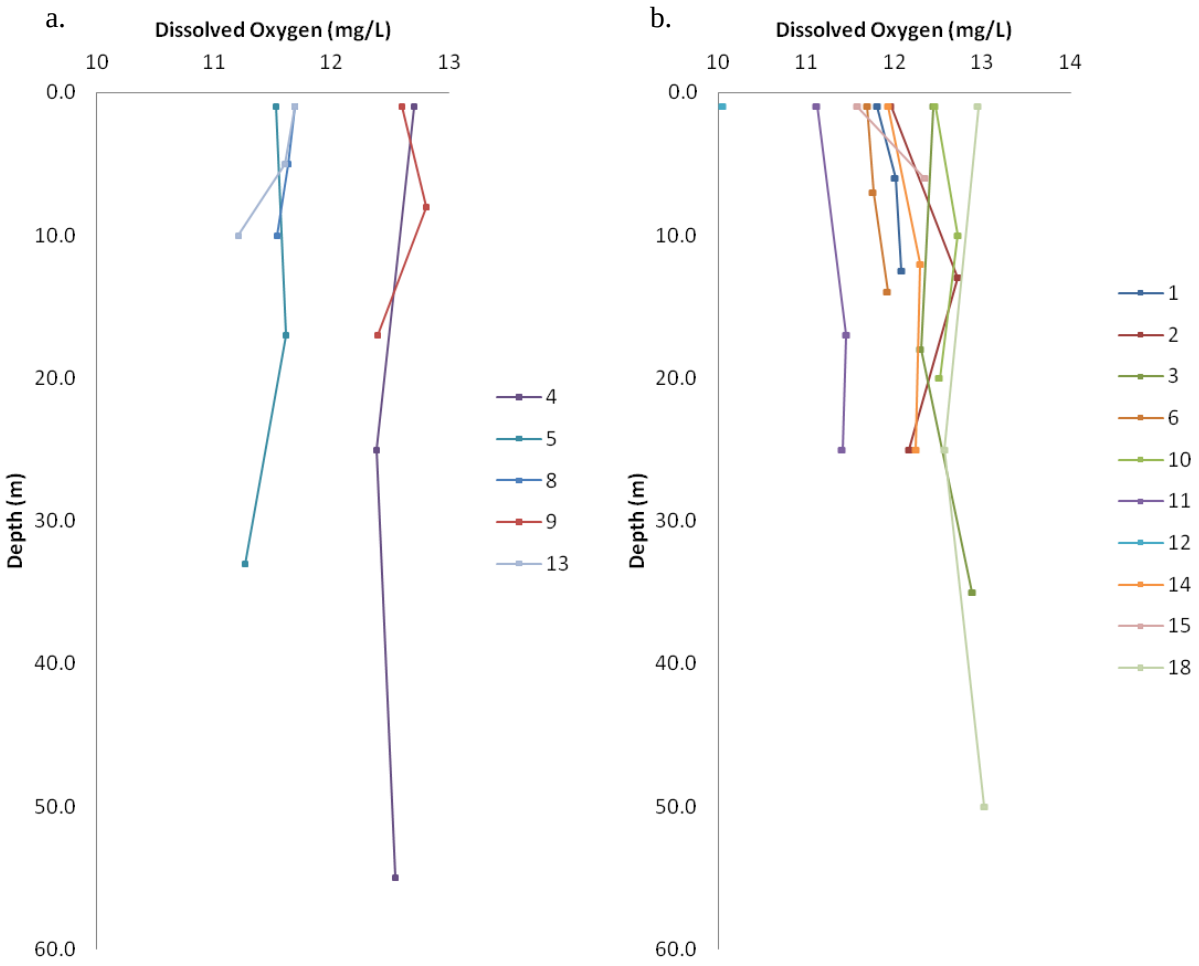


Figure 22. Depth profiles of dissolved oxygen at all sites offshore from Scott Base, with stratification trends showing the notable decrease (a) and increase (b) in DO.

3.2.5 Biological Oxygen Demand

Biological oxygen demand was measured in wastewater samples collected both before and after treatment. The pre- treatment sample had a BOD_5 value of 20 mg/L, and the value for the post- treatment sample was 6 mg/L.

BOD was also measured at site 14, directly offshore from the outfall. Upon reaching the environment, BOD_5 increased from 6 to 12-14mg/L at site 14 (Table 16,) and further from the outfall, at the RO intake, the BOD was even higher at 24 mg/L.

Table 16. BOD_5 concentrations from seawater and wastewater at Pram Point and Scott Base respectively. The 0 value at site 14(25) is thought to be due to malfunctioning equipment.

Site	BOD_5 mg / L
Pre- Treatment	20
Post- Treatment	6
RO intake	24
14	12
14 (12)	14
14 (25)	0

3.2.6 Total Organic Carbon

Total organic carbon was sampled at the top of the water column at nine representative sites offshore from Scott Base (Figure 23). Within this subset of nine sites, four sites were selected to measure stratification of TOC in the water column (Figure 24).

Site 15 recorded the highest TOC concentration at a depth of 1m, with 8.7 mg/L. The lowest concentration (1.0 mg/L) was nearest the outfall also at a depth of 1m. Due to insufficient data for other sites, it is difficult to define a distribution plume of TOC. However, with the limited number of sites samples, it appears that at 1m below the sea ice TOC concentrations increase with distance from the outfall. Within 38m of the outfall, a cluster of sites had TOC concentrations ranging from 1.0 mg/L to 1.8 mg/L. Extending beyond this cluster, TOC concentrations increase to 2.2 mg/L at site 8, and 3.0 mg/L at site 1. The exception is site 15, which recorded the highest TOC concentration of 8.7 mg/L.

The concentrations of TOC in samples collected 1m below the sea ice had no significant correlation with distance from outfall and TOC concentration. Likewise with the limited data at the middle and bottom depths, there was no significant correlation between TOC concentration and distance from the outfall.

Stratification was investigated at four sites extending out from the wastewater outfall. At three sites (1, 8, and 13), TOC was highest at 1m below the sea ice, and concentrations decreased with depth. At the remaining site that was nearest the outfall (site 14), TOC concentration increased slightly with depth (Figure 24).

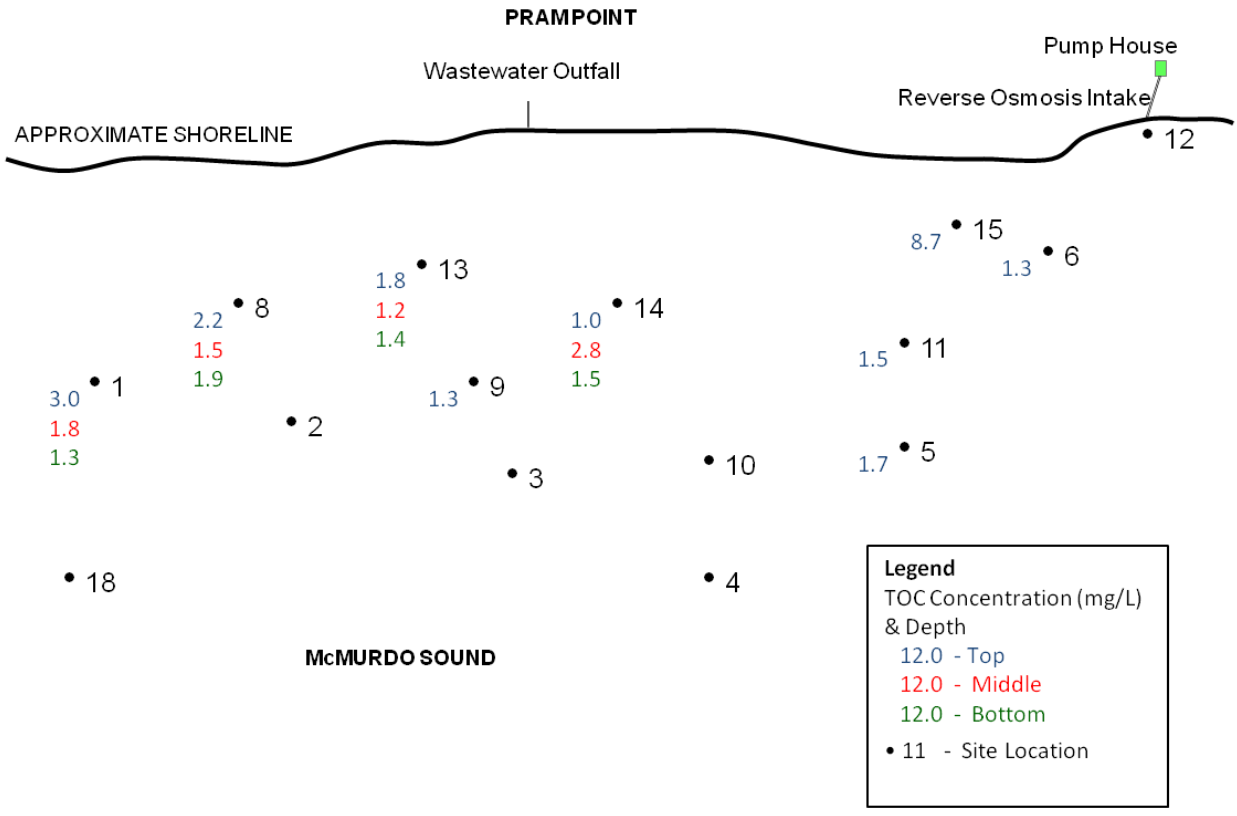


Figure 23. Total organic carbon concentrations at representative sites offshore from Scott Base. Note that depth profiles were measured at sites 1, 8, 13 and 14. At sites 5, 6, 9, 11 and 15 samples were only taken at 1m below the sea ice. All other sites were not sampled.

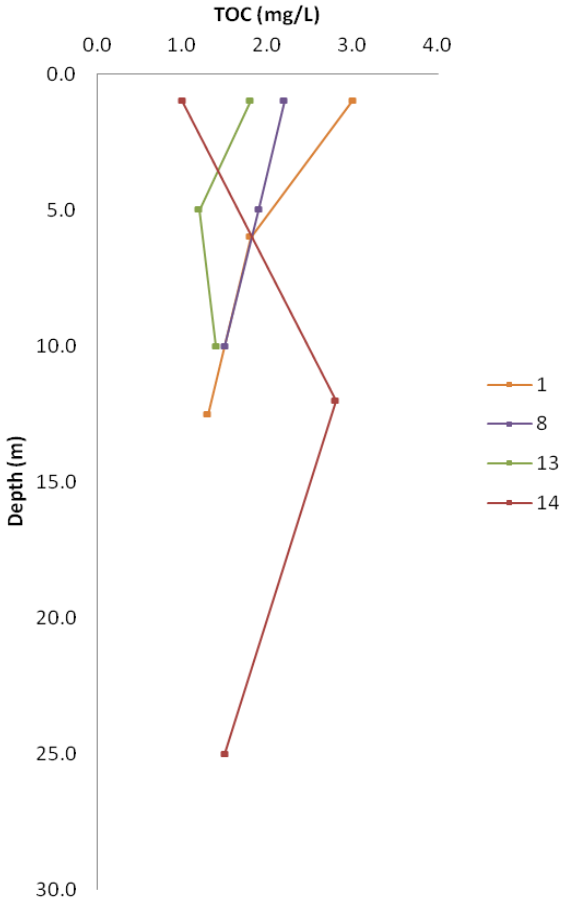


Figure 24. Depth profiles of TOC, at four sites offshore from Scott Base.

3.2.7 Conductivity

The conductivity of the wastewater post-treatment was measured on two occasions prior to discharge. The conductivity was found to be 2.56 mS/cm on the 10/11/2010, and increased slightly to 3.01 mS/cm on the 20/11/2010.

There appears to be no pattern to the dispersion of conductivity offshore from Scott Base at all three depths (Figure 25). However, if the average of all three depths at a site are taken, a clearer pattern is apparent. The majority of all sites have an average conductivity between 50.4 and 53.5 mS/cm, with the greatest average concentrations in the vicinity of the outfall.

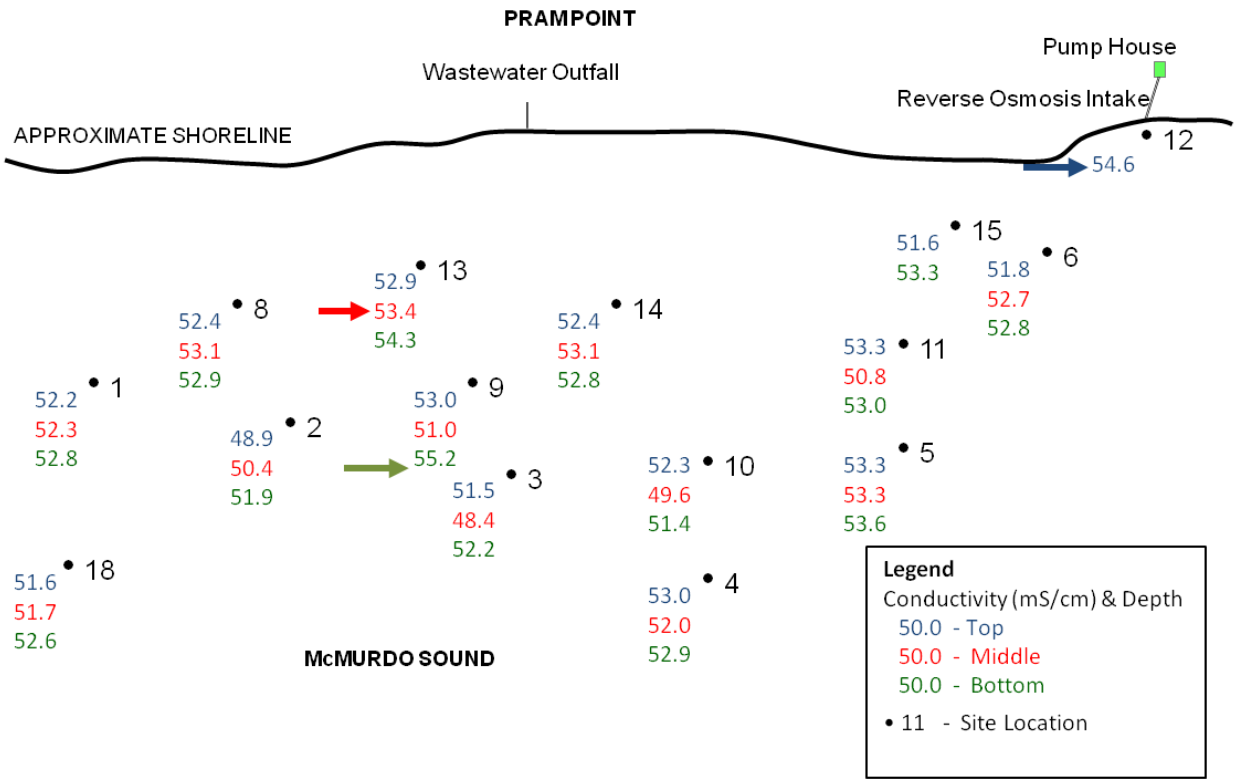


Figure 25. Conductivity at each depth with site locations in relation to WWTP outfall and RO intake. Coloured arrows indicate the greatest concentrations of conductivity for each depth.

For all three depth profiles, there was no significant correlation between conductivity and the distance from the outfall, and stratification of conductivity is relatively variable (Figure 26). At sites 1, 5 and 18 conductivity is uniform with depth with only 1 mS/cm difference between the top and bottom of the water column, while sites 2, 13 and 15 have consistent increases of conductivity with depth. At the remaining sites, there is a notable influence on the middle of the water column with conductivity concentrations increasing at sites 6, 8 and 14. By contrast, there is a slight decrease in conductivity in the middle of the water column at sites 3, 4, 9 10 and 11 (Figure 26).

Conductivity of seawater samples was measured at three sites at Winter Quarters Bay (Table 17). The average conductivity value (51.2 mS/cm) for samples collected Winter Quarters Bay was lower than at Scott Base with. Consistent with other water quality parameters, the average conductivity was greater at the control site with an average of 52.7 mS/cm.

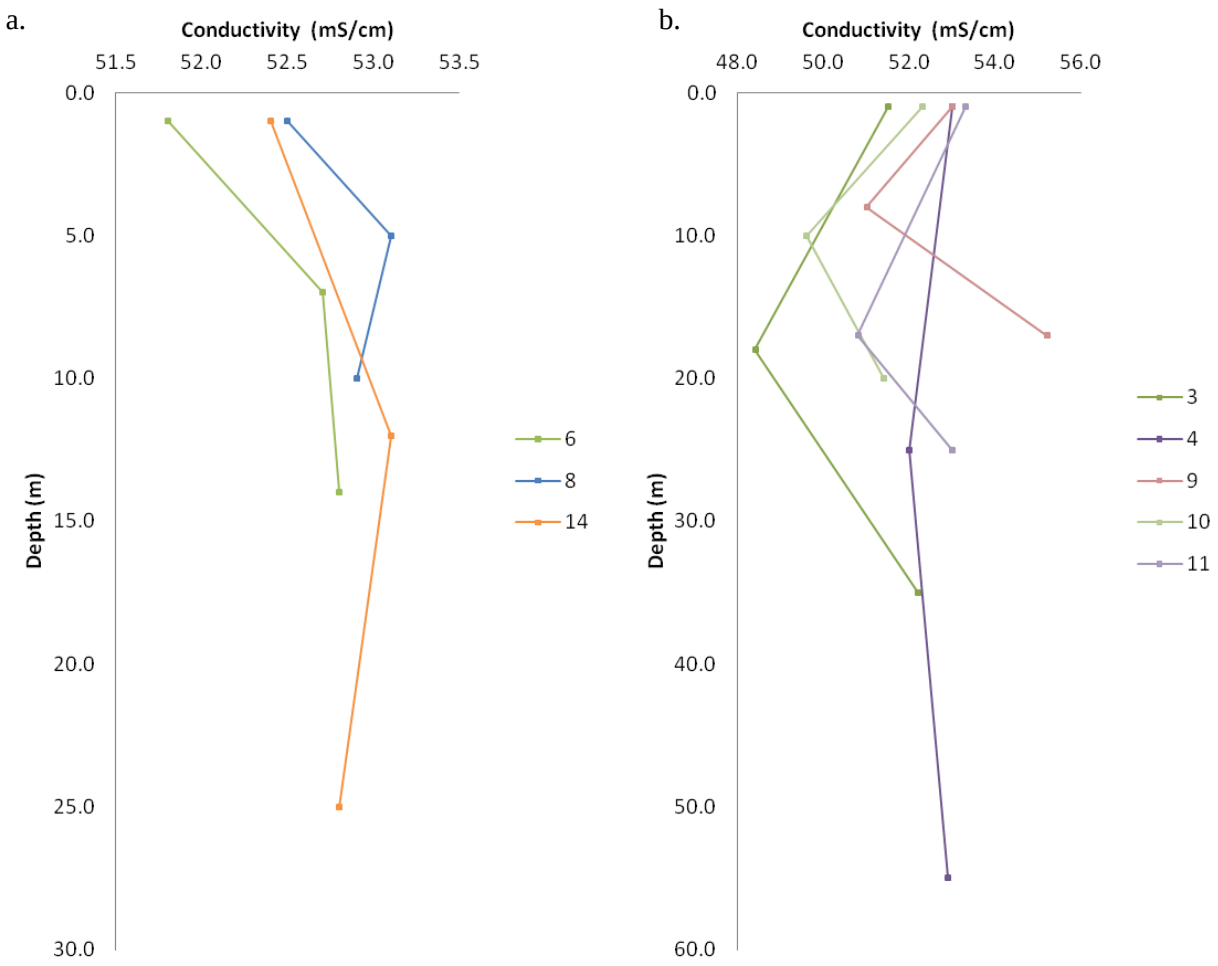


Figure 26. Depth profiles of conductivity, with stratification trends showing the notable increase (a) and decrease (b) in conductivity in samples from the middle of the water column.

Table 17. Conductivity of seawater sampled collected at Winter Quarters Bay, McMurdo Sound.

Site	Depth (m)	Conductivity (mS/cm)
1	1	49.8
1	2	50.7
2	1	51.2
2	9	51.9
2	18	51.3
3	1	52.4
3	15	51.8
3	30	51.5

3.2.8 Total Suspended Solids

Total suspended solid concentrations of pre- and post-treatment wastewater were measured on two occasions (Table 18). On the 10/11/2010, TSS was reduced by 1.6 mg/L between pre- and post-treatment. However, this result is within the error tolerance of 10%, it should be treated with caution. On the 20/11/2010, the TSS reduction was even greater at 6.8 mg/L.

Table 18. Total suspended solid concentrations of pre- and post- treatment samples from Scott Base.

Sample Date	Pre-Treatment (mg/L)	Post-Treatment (mg/L)
10-11-2010	12.4	10.8
20-11-2010	18.4	11.6

The average concentration of all seawater samples recorded was 12.62 mg/L, with a minimum value of 8.69 mg/L and maximum of 17.18 mg/L. Once again, data analysis highlighted what sites recorded the greatest TSS concentrations, and at what depth (Figure 27). Again, there was no clear pattern of dispersion, as the greatest TSS concentrations at each depth occurred at different sites.

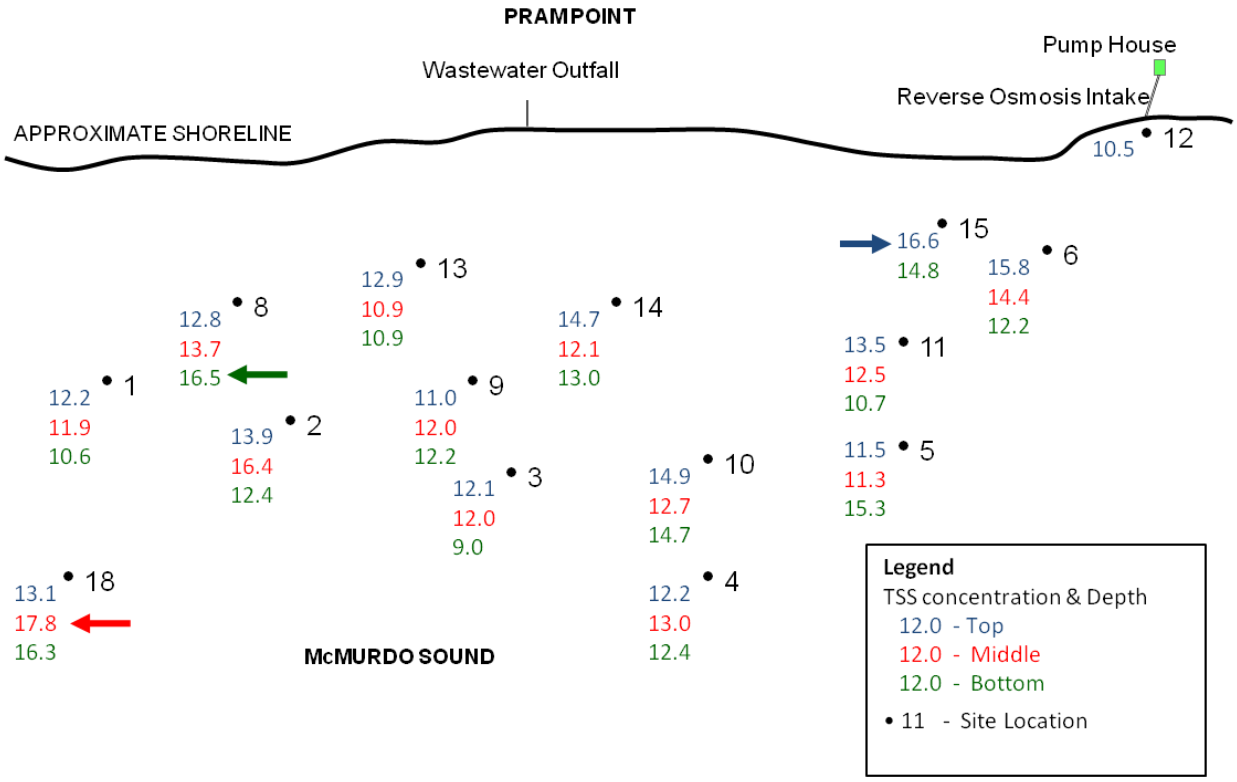


Figure 27. Total suspended solid concentrations at each depth with site locations in relation to WWTP outfall and RO intake. Coloured arrows indicate the greatest concentrations of TSS for each depth.

There was no significant correlation between TSS concentration at each depth, and the distance from the outfall. Similarly, there was no significant relationship between the average of all depths at each site and the distance from the outfall.

Total suspended solid concentrations were measured at three depths at all sites except 15 and 12 in order to create concentration profiles in the water column (Figure 28). The majority of the sites sampled were stratified so there was an overall decrease in total suspended solids with increasing depth. The exceptions were sites 4, 5, 8, 9 and 18 which had greater concentrations at the bottom of the water column.

Total suspended solids concentrations from Winter Quarters Bay (Table 19) were similar than those recorded at Scott Base, with an average of 13.6 mg/L. This value also exceeds that observed at the control site which is 11.35 mg/L.

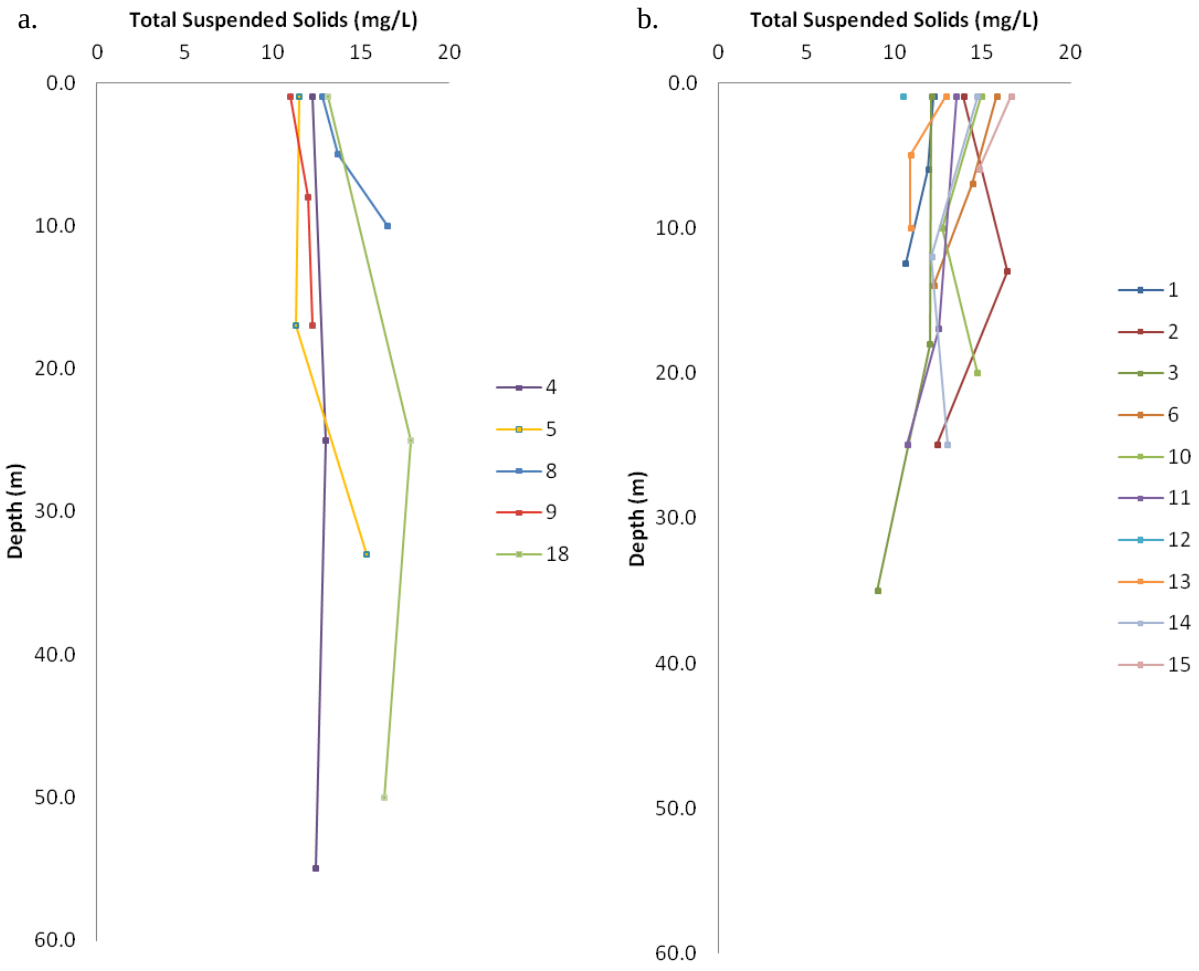


Figure 28. Depth profiles of total suspended solids, with stratification trends showing an increase (a) and a decrease (b) of TSS concentration with increasing depth.

Table 19. Total suspended solid concentrations of sweater samples collected at Winter Quarters Bay, McMurdo Sound

Site	Depth (m)	TSS (mg/L)
1	1	14.6
	2	12
2	1	11.6
	9	12
	18	18.4
3	1	11.4
	15	17.2
	30	12.2

3.2.9 pH

Wastewater post-treatment pH was 7.62 and 7.59 on 10/11/2010 and 20/11/2010 respectively. In the marine environment, there was little difference as pH ranged from 7.42 to 7.92. Within the water column, pH decreased with depth at all sites, except for sites 10, 11 and 14 which are clustered closest to the outfall (Figure 29). However, the difference in pH between the maximum and minimum values at each site was small, ringing between 0.01 and 0.48 pH units.

There was no significant correlation between pH and the distance from the outfall at all depths. At all depths, pH only slightly deceases closer to the outfall. The lowest pH is recorded at site 12 next to the RO intake (Figure 27), as the low pH (7.42) at site 9(17) is believed to be an error in measurement.

The average pH of samples from Winter Quarters Bay (Table 20) was 7.87, which was similar to that observed at Scott Base and the control (pH = 7.80).

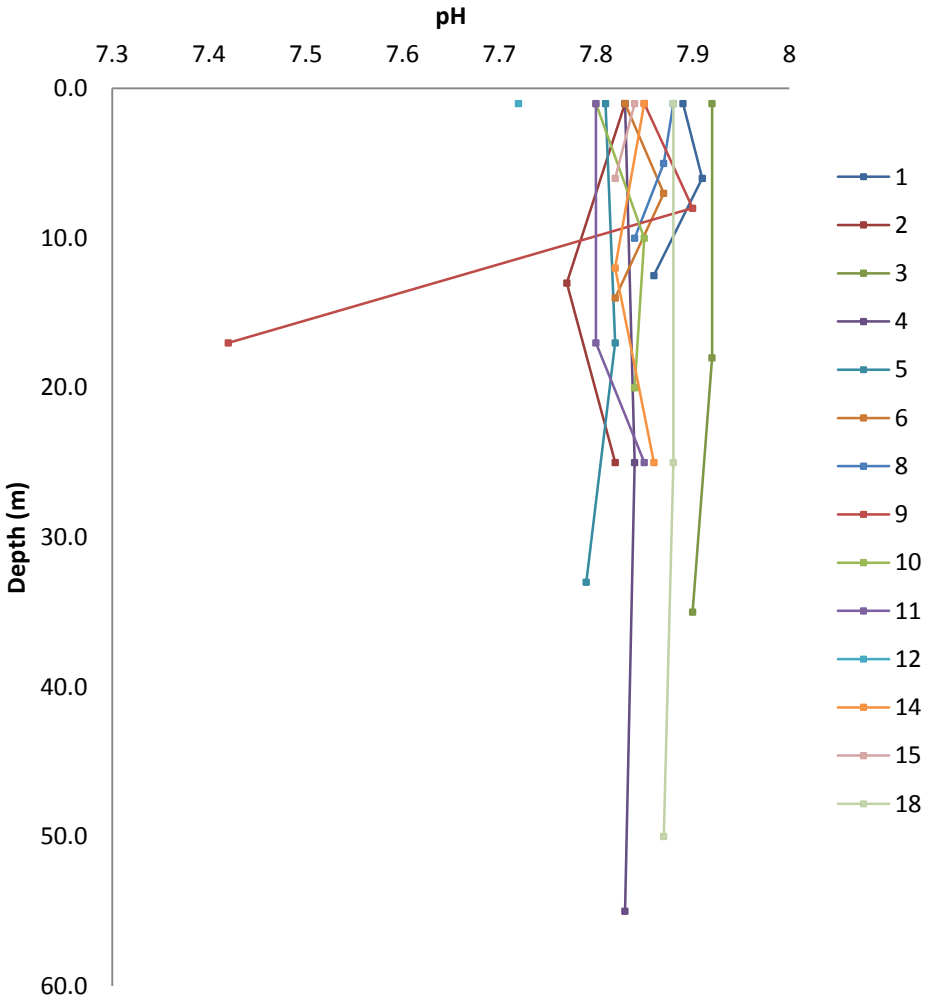


Figure 29. Depth profiles of pH at all sites offshore from Scott Base.

Table 20. pH of seawater samples collected at Winter Quarters Bay, McMurdo Sound.

Site	Depth (m)	pH
1	1	7.88
	2	7.85
2	1	7.86
	9	7.87
	18	7.9
3	1	7.83
	15	7.86
	30	7.84

3.2.10 Turbidity

Turbidity pre-treatment ranged from 47 on the 10/11/2010 to 51 FAU 20/11/2010 (61 to 66 NTU), while post-treatment it was 30 to 48 FAU on the 10/11/2010 and 20/11/2010 respectively (39 to 63 NTU). The turbidity in the water column at Scott Base, Winter Quarters Bay and the control site remained uniform across all sites with a value of 0 FAU. At the RO intake it was also 0 FAU.

3.3 Discussion

This section draws together results from the present study, and compares them to previous studies conducted at Scott Base, and studies at other Antarctica Research Stations. Each parameter is also be assessed for its suitability as a wastewater indicator in the Antarctic marine environment to define the position of the plume.

3.3.1 Faecal coliform and *Escherichia coli*

This study found that although there were high counts of FC and *E. coli* immediately post-treatment, no bacteria were present in the local marine environment. These data are in partial agreement with the most recent routine monitoring FC data of Scott Base's WWTP by Antarctica New Zealand (2011). The routine monitoring determined FC counts in samples from the RO intake, and in the WWTP wastewater after final treatment, prior to being discharged to sea. Monitoring data from Antarctica New Zealand between April 2009 and Janurary 2012 shows high counts of FC's at the WWTP (1000 – 874000 CFU /100ml), with several occasions recording FC to be 'too numerous to count'. These high counts are generally recorded in April, coinciding with the end of the summer field season. At the RO intake, FC have rarely been recorded. On occasions that have detected FC, it has been during December - April, and concentrations have ranged from 1 to 1000 CFU /100ml. The Scott Base population is also recorded at the time of sampling, but there appears to be little relationship between number of people on base and FC in the WWTP treated effluent.

Prior to the installation of wastewater treatment at Scott Base, Battershill (1992) found no contamination in offshore seawater samples. However, routine monitoring by Antarctica New Zealand (1999) had shown FC concentrations up to 2250 CFU /100 ml at the RO intake. Redvers (2000) found maximum FC concentrations at the RO intake to be 90 - 100 CFU /100 ml, and maximum FC concentrations in the offshore marine water of up to 3550 CFU /100 ml in 1998 and 2700 CFU /100 ml in 1999. Redvers (2000) also noted that the concentration and extent of FC was slightly greater on the outgoing tide. At this time, Redvers (2000) concluded that the relocation of the outfall in 1999 reduced the FC contamination of the seawater offshore from Scott Base.

Therefore, the relocation of the outfall and the installation of the WWTP has reduced contamination significantly. The installation of the WWTP has reduced the FC concentration in the wastewater being discharged into the marine environment from an average of 6512250 CFU /100 ml in 1999 to 277500 CFU/ 100ml in 2010, yet the FC target of 200 CFU/100 ml is exceeded on many occasions. In the marine environment, the result is clear with a complete elimination of FC.

Previous research at other Antarctic research stations have also used faecal bacteria to measure the extent of untreated sewage plumes as they have been found to persist in water and sediments (Bruni *et al.*, 1997; Howington *et al.*, 1992; Hughes and Blenkharn, 2003; Hughes and Thompson, 2004; McFeters *et al.*, 1993). Faecal coliforms have thought to be a good indicator of wastewater pollution as it is assumed that

human pathogens occur with the presence of FCs. Untreated sewage at Dumont d'Urville station resulted in high densities (1000 FC/100ml) in the immediate vicinity of the outfall, dropping to low levels within 2km (Delille and Delille, 2000). Likewise at Rothera Station, elevated FC concentrations from untreated sewage extended up to 575m from the outfall, and more than 800m east (Hughes, 2004). Following the installation of the WWTP at Rothera, FC's were not detected more than 50m from the outfall at Rothera. At Casey Station, after sewage treatment, faecal microorganisms were detected approximately 200m from the outfall (Morris *et al.*, 2000).

At McMurdo Station, high concentrations of FC's have been observed along ~1km of shoreline, extending 200 – 300m offshore. Howington *et al.* (1992) found the highest FC concentration (>1000/ $100ml^{-1}$) in a narrow band within 100m of the outfall, reducing to <100/$100ml^{-1}$ at the farthest extent of the plume. Lisle *et al.* (2004) found similar patterns, with the greatest FC concentrations nearest the outfall, reducing to lower concentrations at the drinking water intake (Table 21). Both of these studies highlighted the need for the RO plant to be working effectively at all times, as contamination of the drinking water was possible. As no data has been published on FC concentrations post treatment since the installation of the WWTP at McMurdo, it is currently unknown if faecal bacteria are still present in the water column at the RO intake.

Table 21. Microbial indicator occurrence in water samples collected at McMurdo Station and local areas (Lisle *et al.*, 2004). Values are expressed as FC $100ml^{-1}$. Geometric means are listed with the range of values in brackets.

Sample Site	No. of Samples	Faecal coliforms	*E. Coli*
McMurdo Sewage Outfall	4	21 (2 - 1.40 x 10^2)	15 (1 - 43)
Intake (Pre-RO)	2	8 (3 - 16)	2 (0 - 5)
Transition South	2	1.37 x 10^2	49 (10 - 87)
Big Razor Back Island	1	0	0
Little Razor Back Island	1	0	0
Cape Evans	1	0	0

The present study has found no FC or *E. coli* in the marine environment offshore from Scott Base, which brings into question where FC can originate from, what factors affect survival and the suitability of faecal coliforms as an indicator of wastewater in Antarctica. Faecal coliforms are generated in Antarctica by two predominant sources; vertebrate populations and human sewage (Hughes, 2003). There are many factors that influence the survival and distribution of faecal microorganisms in the Antarctic marine environment, with survival rates varying from minutes to several days, depending upon the environmental conditions (Statham and McMeekin, 1994). Coliform "die off" can be accelerated various conditions including solar radiation, water salinity, temperature, sea ice conditions, nutrient limitation, predation and anthropogenic input (Hughes, 2003; Statham and McMeekin, 1994). It has been recognised that solar radiation may be the most significant limiting factor in Antarctic marine environments, and during the austral summer high doses of Antarctic solar radiation may cause inactivation of bacteria over

short periods of time (<1h) (Hughes, 2005). In an earlier study, Hughes (2003) found that during the month of February high solar radiation doses resulted in low FC counts, but by April radiation doses had reduced by ~95% and FC concentrations increased by approximately 1000-fold.

The formation of sea ice in September also increases FC survival despite ozone depletion and moderate solar radiation (Hughes, 2003; Statham and McMeekin, 1994). Despite the reduction of radiation and increased sea ice promoting the survival of FC concentrations, the timing of these factors also coincides with a decrease in Base populations which would ultimately reduce the impact of bacteria in the water column. Hughes (2005) also investigated the viability of *E. Coli* in seawater under radiation to find that cell survival was less in seawater, which may be linked to osmotic stress rather than doses of radiation. However, at Scott Base it is common for sea ice to be fast against Pram Point during the summer, as it rarely breaks out. Coupled with the increased population at Scott Base, FC from the wastewater could persist much longer as the effluent is discharged under the sea ice and protected from radiation. Despite conditions (increased sea ice cover and discharge during the summer) at Scott Base being favourable for the survival of sewage bacteria, no FC or *E. coli* were observed in this study. This may be due to a combination of the wastewater treatment, sufficient mixing by local currents, and/or the nature of discharge. As the discharge from Scott Base is a constant low volume flow, the potential protective effect of increased turbidity in the water column is unlikely to be the cause (Statham and McMeekin, 1994).

Recommendations

Due to the variability of natural environmental factors (Hughes, 2003), the use of microbial indicators may not be a suitable parameter to measure wastewater discharges in Antarctica. Therefore FC cannot be reliably used to locate the position of the plume for the sewage outfall at Scott Base. However, FC and *E. coli* should be monitored at site 12, or continued to be monitored at the RO intake as the implications for human health are severe if drinking water is contaminated by sewage pathogens (the implications for human health will be discussed in more detail in chapter four). It is recommended that *Streptococus feacalis* or *Enterococci* also be used as an indicator of wastewater pollution, as it more resistant to environmental stress and is closely associated with human and animal faeces (Owili, 2003).

3.3.2 Trace Metals

To assess concentrations of trace metals in the marine environment, an understanding of trace metals in wastewater discharge must first be investigated by comparing trace metal concentrations in samples collected before and after wastewater treatment. Results indicate that the concentrations of most metals are increased after wastewater treatment. On the first sampling occasion, there was up to a 43.1% increase in total and dissolved concentrations of Cu, Fe, Ni and Mn. By comparison in the second sampling round, total and dissolved concentrations of Cu, Ni, and Zn showed up to a 23.8% increase after treatment. Concentrations of Fe and Mn were reduced up to 15.9% in both total and dissolved samples.

Results indicate that trace metals are increased during wastewater treatment, yet what is of importance for this study is if the elevated concentrations are also observed in the marine environment offshore from Pram Point. When concentrations of post-treatment samples are compared to marine water samples, there is a reduction in trace metal concentrations for a vast majority of sites sampled. Concentrations of total and dissolved Fe, Ni and Mn decreased by an order of magnitude with Zn and Cu decreasing by approximately 50% for both total and dissolved concentrations. There was only one sample to increase in trace metal concentrations between post-treatment wastewater and marine water. This was at site 15 where total Cu concentrations increased from 8.52 and 6.82 µg/L in post-treatment samples to 26.49 µg/L in marine samples, and dissolved concentrations in the marine environment were 22.23 µg/L compared to post-treatment values of 6.00 and 7.23 µg/L.

Despite trace metal concentrations generally being reduced in the marine environment compared to treated wastewater, the marine water data highlights metals that may be an indicator of the wastewater plume. Total and dissolved concentrations of Cu and Zn are only elevated at sites 13, 14 and 15, which are closest to the shore and clustered around the outfall. Similarly, Mn concentrations were greatest at sites 13, 14 and 15, with other sites offshore from Pram Point recording concentrations that were not elevated. Total concentrations of Fe were observed at the majority of all sites sampled, and greatest concentrations were observed at site 14 and 15, while only one dissolved sample from site 13 was measureable. At all the sites sampled, total and dissolved concentrations of Ni were detected, with the greatest concentrations once again being at sites 13 and 15.

Stratification can only be investigated for concentrations of Ni in the water column due to much of the trace metal data being below detection limits. It was expected that for a buoyant plume, concentrations would decrease with depth at sites near the outfall. However, it was found that there was no clear stratification at sites 4, 9, 13, 14 and 18 for total and dissolved concentrations of Ni. For example, total concentrations were greatest at the top of the plume at site 14 (closest to the outfall), but also at site 18 (farthest from the outfall).

The metal contamination in seawater at Scott Base has been compared to those of Redvers (2000) study and other coastal Antarctic waters. Redvers (2000) collected representative surface samples from six sites near shore to Pram Point and used graphite furnace - atomic absorbance spectrometry to analyse for Cu and Zn (Table 22). Redvers (2000) found that Cu and Zn concentrations were greater near the wastewater outfall from Scott Base. As site locations changed between 1998 and this study, and some data from the present study is below the detection limit, only one site offers a direct comparison with Redvers (2000). This comparable data is from the 1998 site 16 and the present study's site 13. The Cu concentrations measured by Redvers at site 16 in 1998 were 1.2 µg/L, while this study found greater concentrations of 2.1 µg/L (total) and 3.1 (dissolved). There was a greater increase of Zn, as Redvers (2000) recorded concentrations of 2.4, while only a total concentration of 35.9 µg/L was measured in this study.

Table 22. Marine water trace metal concentrations sampled in 1998 at a depth of 1m from Redvers (2000).

1998 Site	Cu (µg/L)	Zn (µg/L)
4	0.4	2.0
9	0.5	1.0
12	0.8	1.1
13	0.6	1.7
15	3.2	4.8
16	1.2	2.4

The observed increased of Cu and Zn concentrations between Redvers (2000) and the present study could be due to the location of the outfall. In 1998 the wastewater was being discharged to land, and the subsequent relocation of the outfall in 1999 to discharge directly into the sea may have resulted in greater trace metal concentrations being observed in the present day.

Currently no wastewater standards apply to anthropogenic input of trace metals in the Antarctic marine environment. However, when assessing the impact of wastewater discharge from Scott Base on the marine environment, it is useful to compare the data to other acceptable National water quality guidelines. The Australian and New Zealand Environment Conservation Council (ANZECC) water quality guidelines for marine water (ANZECC, 2000) for example give trigger values for marine ecosystem protection. All detectable levels of Cu and Zn offshore from Scott Base are above the current trigger values for marine water Cu and Zn (Table 23). There is insufficient data to derive an ANZECC trigger value for Fe, Mn or Ni. In Antarctica, a 99% level of protection should be aimed for, yet this cannot be confirmed for Cu and Zn as the detection limits used for this study are above the trigger values. However, this study can confirm that a 95% level of protection is being met at the majority of sites and depths for concentrations of Cu and Zn collected in marine water samples offshore from Pram Point. Total and dissolved Cu concentrations for sites 4(25), 13(1) and 14(12) are such that a 95% of species protection is expected, whilst site 15(6) is reduced to an expected species protection of 80%. Total Zn concentrations exceed the 90% trigger value at site 13(1), with both total and dissolved concentrations at site 15(6) exceeding the 90% species protection.

Table 23. Trigger values for metals at alternative levels of protection (ANZECC, 2000).

Trace Metal	**Trigger Values (µg/L)**				**Total Concentration (µg/L)**				**Dissolved Concentration (µg/L)**			
	Level of Protection				Site (Depth in Metres)				Site (Depth in Metres)			
	99%	95%	90%	80%	4 (25)	13 (1)	14 (12)	15 (6)	4 (25)	13 (1)	14 (12)	15 (6)
Cu	0.3	1.3	3	8	2.67	2.10	2.54	26.49	< 1.2	3.14	< 1.2	22.23
Zn	7	15*	23	43	< 12	34.92	< 12	27.18	< 12	< 12	< 12	28.45

*Figure has been outlined as a trigger value that may not protect key test species from chronic toxicity.

The United States Environmental Protection Agency (EPA) also publishes National Recommended Water Quality Criteria (EPA, 2009), which measures pollutants against acute and chronic toxicity levels in saltwater. Under these criteria, total and dissolved Cu concentrations at site 15(15) exceed acute levels of 4.8 µg/L. Likewise in the United Kingdom, environmental quality standards are derived under the requirements of the Dangerous Substances Directive (76/464/EEC). Once again the total and dissolved Cu concentration at site 15(15) exceed the annual dissolved standard of 5 µg/L.

The sites which have exceeded trigger values for toxicity to marine organisms are once again located near the shore, and clustered around the outfall. However, due to the sparsely populated habitat at these sites (Battershill, 1992) there is minimal risk of Cu and Zn toxicity to benthic invertebrates. The only real risk could be to mobile vertebrates and invertebrates, yet the doses are so low that significant periods of time would need to be spent near the outfall to be toxic (Evans *et al.*, 2000).

Elevated levels of introduced trace metals in the marine environments have been historically linked to numerous anthropogenic sources, including shipping activity, sewage outfalls, antifouling paints and station operation (Curtosi *et al.*, 2010). Other sources in Antarctica also include atmospheric input from industrialised and urban areas in the Southern Hemisphere (Sanchez - Hernandez, 2000). Monitoring of trace metal contamination in the Antarctic marine environment has highlighted several sources of contamination (Conlan *et al.*, 2004; Cunningham *et al.*, 2005; H. S. Lenihan and Oliver, 1995; Negri *et al.*, 2006; Townsend and Snape, 2008). Elevated levels of Cu and Zn in sediment have been measured in areas of shipping activity, and it is possible that these metals have come from antifoulant paints (Negri *et al.*, 2006) or from Zn coatings that protect iron and steel. Additionally, sources of Pb are thought to be paint and fuel spills with batteries the dominant sources overall (Townsend and Snape, 2008).

Data collected on trace metals in marine seawater (Papoff *et al.*, 1996; Westerlund and Ohman, 1991), sediments (Green and Nichols, 1995; Montone *et al.*, 2010; Ribeiro *et al.*, 2011; Townsend and Snape, 2008) and organisms (Curtosi *et al.*, 2010; Negri *et al.*, 2006; Stark *et al.*, 2003a) has generally concluded that anthropogenic influences of trace metals on coastal environments in Antarctica are negligible. As investigation of the effects in the seawater is often difficult, sediments have more commonly been used to measure contamination in the Antarctic marine environment. For example, seawater monitoring conducted by Crockett (1997) near McMurdo Station yielded unreliable results, and the study reverted to investigating metal concentrations in sediments near the outfall. Crockett's (1997) results indicate that the sediments near the wastewater outfall contain high concentrations of Cu, Zn and Pb and were linked to elevated levels of the same metals in effluent. Additionally, it has been found that increased concentrations nearest the shore may also arise from overland transportation of base soils. Metal concentrations in sediment have been found to be 1000 - 100,000 times higher than in the above water and often show little spatial and temporal variation (Bargagli, 2000).The potential of trace metals leaching from soil into the marine environment at Scott Base will be discussed in chapter four.

However, some trace metal concentrations have reduced due to a change in anthropogenic behaviour. For example, Pb has been recognised as being mainly of anthropogenic origin in Antarctica, and has reduced in recent years due to the diminishing use of leaded fuel use in Southern Hemisphere countries (Sanchez - Hernandez, 2000). Concentrations of dissolved trace metals can be also affected by the presence of ice cover, algae and phytoplankton blooms in summer. The seasonal variation in Cd was investigated at Terra Nova Bay, and it was found to be significantly depleted once the sea ice melted and phytoplankton bloomed. Therefore, this seasonal effect of light and ice cover in Antarctica, together with seawater upwelling should be considered when comparing data from different regions, or even the same region but different times (Bargagli, 2000). In this study, an effort was made to sample at the same period of the tide as Redvers (2000), and at the same time of year. Sea ice conditions were also similar with total cover during sampling.

Recommendations

Despite the difficultly of analysing for ultra-trace levels of metals in the marine environment, metals such as Cu and Zn could be used to measure not only wastewater contamination, but also contamination from historical activities at Scott Base. The availability of water quality guidelines from outside of Antarctica also provides a partial benchmark for monitoring, but highlights the need for water quality guidelines for Antarctica itself.

3.3.3 Nutrients

It was expected that NO_3^-- N and PO_4^{3-} concentrations would be reduced upon reaching the marine environment, yet elevated at sites closest to the outfall, and at the top of the water column due to the buoyant characteristics of the wastewater plume (Redvers, 2000). The levels of NO_3^-- N and PO_4^{3-} in samples collected before and after the wastewater treatment plant were consistently greater than 5.5 mg/L, yet were greatly reduced in the marine environment. The greatest NO_3^-- N concentrations of 1.1 mg/L were recorded at site 9 and 14. However, the sample from site 9 was taken at the bottom of the water column, and from 1m below the sea ice at site 14. Similarly, the greatest PO_4^{3-} concentration was 0.8 mg/L at site 15, located 24m from the outfall, at a depth of 6m. This indicates that a localised plume exists. However, Pearson correlation analysis confirmed that there was no significant relationship between distance from the outfall, depth and PO_4^{3-} or NO_3^-- N.

The buoyancy of the wastewater plume was then investigated by using stratification data to determine how the concentrations were distributed in the water column. A buoyant plume is expected to have greater concentrations at the top of the water column than at the bottom. Stratification of NO_3^-- N in the water column was difficult to determine as only three samples recorded NO_3^-- N concentrations over the estimated detection limit of 0.8 mg/L. Subsequently, there can be little speculation as to how this parameter is distributed vertically in the water column. By contrast, stratification of PO_4^{3-} was investigated more successfully as all marine water samples analysed for PO_4^{3-} recorded concentrations

above the detection limit of 0.05 mg/L. However, there is little evidence of stratification due to a random distribution values within the water column, and some sites having greater concentrations at depth. This trend may be due to dead biota and faecal pellets sinking and dissolving down the water column (Westerlund and Ohman, 1991), or the mixing effect of the tide.

To obtain a more thorough understanding of the extent of the plume, an average of PO_4^{3-} concentrations from all three depths at a site was calculated. This produced a clear pattern of dispersion (Figure 18), with concentrations decreasing with distance from the outfall. This suggests that the extent of the plume is localised, yet the water column is not stratified.

In other studies, Crockett (1997) found that total P (0.06 - 0.09 mg/L) and NO_3^-- N (0.38 - 0.43 mg/L) at McMurdo were not elevated within the vicinity of the wastewater discharge. Westerlund (1991) reported PO_4^{3-} concentrations of ~0.2 mg/L in surface waters of the Weddell Sea. At Scott Base, Redvers (2000) found the greatest concentrations of total N and total P occurred nearest the outfall, at depth of 1m below the bottom of the sea ice, and on the incoming, rather than the outgoing tide. As Redvers (2000) measured total nutrients and the present study measured NO_3^-- N and PO_4^{3-} , no direct comparison is possible between the actual values measured in each study. However, in this study we can compare where the plume signal was strongest.

Redvers (2000) observed that the plume signal was strongest at sites closest to the outfall, and also that the plume increased in the concentration and spatial extent between 1998 (max. total N = 2.20 mg/L, total P = 0.52 mg/L) and 1999 (max. total N = 10.88 mg/L, total P = 0.83 mg/L). This was thought to be due the outfall being relocated so that the wastewater was being discharged directly into the sea. The sites that showed the highest concentrations were consistently closest to the outfall. By comparison of this current study, the maximum NO_3^-- N concentrations was 1.1 mg/L at site 9 and 14 , and the maximum PO_4^{3-} concentration of 0.80 mg/L was measured at site 15. Comparing the two studies it be concluded that the highest concentrations are mainly occuring at the same sites.

It is apparent from further investigation and analysis of data collected by Redvers (2000) in 1999 that on the incoming tide there was clear stratification of nutrient levels with greater concentrations at the top of the water column. However, on the outgoing tide there was a reversal in stratification of total N at many sites (8, 9, 10, 11, and 13) with higher concentrations observed at depth. Total P concentrations observed by Redvers (2000) between the tide phases had no reversal in stratification except at one site (15). Comparison of the PO_4^{3-} stratification in the water profiles of this study reveals that nothing has changed.

However, the maximum nutrient concentration does not effectively provide a good assessment of the impact of the WWTP discharge. By looking at the distance from the outfall that PO_4^{3-} concentrations return to background levels being reached, it is clear that the plume now only extents up to 30m offshore and 35m longshore, a noteworthy reduction since Redvers (2000). Overall, this conclusion illustrates that

despite the relocated outfall discharging directly to sea, the improved wastewater treatment has had a measurable improvement in the marine environment.

No guidelines exist for nutrient concentrations in Antarctica seawater, or any other specific guideline provided by any other national standards. The ANZECC guidelines have insufficient data available for a nitrate trigger value in seawater, and there are no guidelines for phosphate.

Recommendations

In this study, NO_3^-- N and PO_4^{3-} were not good indicators of the plume because the plume is restricted to a very localised area and the concentrations were generally low, or below the detection limit of the apparatus. However, nutrients should be continued to be monitored because they are prevalent in wastewater, and can cause significant degradation to the environment. The addition of NO_3^-- N and PO_4^{3-} into the naturally nutrient-limited waters of Antarctica can cause increased abundance of microbial organisms, and contribute to eutrophication. This study has confirmed that NO_3^-- N is greatly reduced as a result of wastewater treatment, and PO_4^{3-} is still detectable, but at low concentrations. Future monitoring should ensure that the WWTP continues to reduce the impact of nutrients in the marine environment offshore from Pram Point.

3.3.4 Dissolved Oxygen, Biological Oxygen Demand and Total Organic Carbon

Monitoring of dissolved oxygen determines the amount of oxygen available for organisms in the water column and biochemical oxygen demand measures the amount of oxygen consumed by microorganisms in decomposing organic matter. Additionally, total organic carbon assesses the organic load in the water. It is useful to investigate these three parameters together as BOD and TOC directly affect the amount of DO in the environment. In general, an increase in BOD and TOC results in the amount of DO rapidly decreasing.

Dissolved oxygen

Dissolved oxygen concentrations pre-treatment were 0.09 mg/L and 0.15 mg/L, and post-treatment DO was 20.61 mg/L, and 20.89 mg/L. In the marine environment, the greatest DO concentrations was 13.02 mg/L recorded at site 18, which is located the farthest from the outfall at 62m. The lowest DO concentration was 10.05 mg/L at site 12. This is to be expected, as within approximately 35m of the outfall, pollution stimulates the growth of organic matter that depletes the available oxygen. As the level of pollution decreases away from the outfall, the average concentration of DO is increased at sites. The minimum DO of 10.05 mg/L at site 12 may not be due to microorganisms in the wastewater depleting oxygen, but a result of the increased salinity at this site (Kramer, 1987). Site 12 is the intake for the reverse osmosis plant, and is also where waste brine is returned from desalinisation. Despite the minimum DO concentrations being near the outfall, and increasing in concentration at sites further away, overall there was no significant correlation between DO and the distance to outfall.

When all three depth profiles were investigated, DO was slightly reduced near the outfall at 1m below the sea ice. Overall, the sites were stratified so they had increasing concentrations of DO with depth. This is once again to be expected as there would be more oxygen-demanding organic matter at the top of the water column consuming oxygen.

Comparison was made between all sites of Redvers (2000) and the current study. Redvers (2000) measured DO concentrations between 5.2 mg/L at site 14, and 11.4 mg/L at site 2. The present study ranged from 10.05 mg/L at 12 and 13.02 mg/L at 18. This indicates that DO is higher at sites further from the outfall in both studies. To understand the general magnitude of change in the marine environment resulting from the installed wastewater treatment plant, the mean of all data is compared between Redvers (2000) and the present study. The result is that the mean DO concentration has increased by 2.64 mg/L (Table 24). This is believed to be as a result of improved wastewater quality, with temperature not being a factor as the two studies were conducted at the same time of year.

Table 24. Comparison between average DO results from Redvers (2000) and present study.

Study	DO (mg/L)
Present	12.04
Redvers (2000)	9.40

Redvers (2000) reported that at all sites outside the influence of the plume, DO was all relatively uniform with depth, and at sites closer to the outfall the influence of the plume is apparent. Redvers (2000) stated that DO increased in concentration with depth "for all sampling runs". Yet, this parameter was only measured on the incoming tide. As it has been recognised that there was differences in stratification between the outgoing and incoming tide, the conclusion is somewhat misleading. By comparison at Winter Quarters Bay, Crockett (1997) found small variations in DO to be between 8 and 9 mg/L.

Biochemical Oxygen Demand

The pre- treatment sample had a BOD_5 value of 20 mg/L, and the value for the post- treatment sample was 6 mg/L. This indicates that the wastewater treatment plant is reducing the amount organic matter. The BOD_5 ranged from 12 to14 mg/L at representative sites near the outfall. This increase upon reaching the marine environment may be due less organic matter available for microorganisms to digest. It has been reported that BOD_5 values from sewage that has been effectively treated would have a value of ~20 mg/L and untreated sewage often has BOD_5 values between 200 and 600 mg/L. The pre- and post-treatment data presented here is considerably below these values, which is probably due to the small base population and effective treatment. Routine monitoring of Scott Base's wastewater discharge has recorded maximum detectable BOD_5 concentrations of 49 mg/L, with elevated concentrations generally being in April.

Redvers (2000) observed BOD maximum concentrations of 7.18 mg/L at sites closest to the outfall, to 1.24 mg/L at sites outside the plume area. On average, the BOD concentration in 1998 and 1999 was

3.89, which has increased to an average of 8.7 mg/L during the present study. By comparison at Winter Quarters Bay, Crockett (1997) found small variations in BOD (2.4 mg/L) occurring near the outfall. Despite the small values of BOD, Crockett (1997) observed a decrease with distance from the outfall. Crockett (1997) also reported BOD to be 0.74 mg/L at the drinking water intake.

Effluent guidelines have varied between Antarctic Research Stations (Connor, 2008). Scott Base's target of 30 mg/L for BOD_5 is generally being met. Recorded concentrations of BOD_5 have only exceeded the target of 30 mg/L on two occasions between 15/10/2004 and 12/1/2012 (note that this is not a continuous record, yet in the last two years records have been more constant) (Antarctica New Zealand, 2011). By comparison, the targets at Neumayer and Jubany Bases are less stringent, with BOD of 50 mg/L, (Ahammer *et al.*, 2000). Davis Station is at the other extreme, with BOD_5 targets of 20 mg/L, (Heaton and Paterson, 2003).

The stated BOD_5 targets appear reasonable for Scott Base, but for other seasonal bases in Antarctica it may not be the case. It has previously been reported that BOD removal efficiency summer only stations is often low due to the lengthy start up time of WWTP's (Connor, 2008). At the Indian, Maitri station, BOD removal efficiency was only between 25 to 40%, and thought to be associated with reduction in effluent pH (Ghosh *et al.*, 1997).

Total Organic Carbon

Total organic carbon is used to indicate the presence of organic contaminants that may provide a nutrient source for harmful microbial growth, or degrade the ion-exchange capacity. Offshore from Pram Point, TOC concentrations were greatest at site 15 with 8.7 mg/L, and ranged to a minimum concentration of 1.0 mg/L. Normally, wastewater contains high levels of organic compounds, with TOC concentrations >100 mg/L (Clesceri *et al.*, 1998). By comparison with other marine environments worldwide, TOC has been recorded from 0.8 mg/L to 6.78 mg/L (Emara, 1998; Visco *et al.*, 2005). In Antarctica, TOC concentrations have been observed at Rothera Station, with concentrations ranging from 2.43 to 0.44% (Hughes and Thompson, 2004).

The elevated concentration at site 15 could be due to its close proximity to the shore, and not from outfall as other sites in the expected area of the plume were between 1.0 mg/L and 3.0 mg/L. At all sites that measured depth profiles (sites 1, 8, 13 and 14), stratification was occurring with greater concentrations occurring at the top of the water column.

Redvers (2000) recorded minimum concentrations of TOC once again at sites farthest from the outfall, with concentrations up to 2.0 mg/L at site 15. On average, the mean TOC is greater for the present study with 2.1 mg/L, while Redvers (2000) only recorded a mean of 0.7 mg/L in the marine environment.

Recommendations

It is recommended that BOD and TOC be only measured in the marine environment more thoroughly if there is a documented concern that there is limited oxygen in the seawater due to excess microorganisms being present (Clesceri *et al.*, 1998). This can be achieved by using a portable metre to measure DO that can measure samples immediately, and do not require transportation to professional laboratories for analysis. To determine if BOD and TOC analysis is required, routine monitoring of DO concentrations in the marine environment should be initiated. This will provide a good data set in order to observe if a significant decrease in oxygen levels occurs.

Additionally, it is recommended that TOC be routinely measured in the monthly WWTP monitoring. If the data provides a repeatable empirical relationship between TOC and BOD, then the TOC can be used to estimate the accompanying BOD (Clesceri *et al.*, 1998). In the past, the BOD measured at Scott Base has been inconsistent and at times failed, so this will provide quality assurance and quality control for the laboratory.

3.3.5 Conductivity, Total Suspended Solids, Turbidity and pH

Conductivity

The conductivity was greatest close to the outfall at site 9 with 55.2 mS/cm. The lowest concentrations were observed farthest from the outfall, with a cluster of three sites that are approximately 32 - 38m from the outfall also recording low conductivity values of 48.4 mS/cm to 52.8 mS/cm. When stratification was investigated, the majority of sites showed an increase of conductivity with depth. However, at some sites there was an interaction in the middle of the water column causing conductivity to increase or decrease. At sites 6, 8 and 14 (which are in a transect along the shore), there is an increase of conductivity in the middle of the water column. By contrast, at sites 3, 4, 9, 10 and 11 (which are clustered directly south of the outfall), there is a notable decrease in conductivity.

Redvers (2000) recorded a mean conductivity of 54.6 mS/cm, a maximum of 56.0 mS/cm and minimum of 40.5 mS/cm. This slight increase in conductivity compared to the present study could be a result of the wastewater treatment reducing inorganic ions that may be present in the discharge. However, it is more likely to be due to different methods of measuring conductivity. Redvers (2000) also reported that conductivity at all sites outside the influence of the plume were relatively uniform with depth, and at sites closer to the outfall the influence of the plume was apparent. This is in partial agreement with the present study, with the exception of those sites previously mentioned that have an increase or decrease of conductivity in the middle of the water column.

Total Suspended Solids and Turbidity

The TSS concentration and turbidity essentially measure the same water quality property, which is the particulate matter suspended in the solution. However, TSS is recognised as a better indicator because it measures the weight of particulate matter in the sample.

It is clear that the WWTP slightly improves the clarity of the water stream, with turbidity pre-treatment ranging from 47 to 51 FAU, while post-treatment it was 30 to 48 FAU. Further dilution in the marine environment results in total clarity of the wastewater, with a turbidity of 0 FAU. Despite turbidity measurements indicating no suspended matter in the water column, samples did record measureable concentrations of TSS.

As expected, wastewater treatment decreases concentrations of TSS by 1.6 - 6.8 mg/L. Total suspended solid concentrations in the marine environment ranged from 9.0 mg/L to 17.8 mg/L, with no clear pattern of dispersion, as the greatest TSS concentrations at each depth occurred at different sites. However, stratification of total suspended solid concentrations indicated there was an overall decrease in total suspended solids with increasing depth. These concentrations are elevated above those stated in previous literature, with Harris and Fabris (1979) recording TSS between 0.1 mg/L and 0.542 mg/L in the Antarctic Ocean. The results from Harris and Fabris (1979) are much lower than recorded offshore from Pram Point, yet the pattern of dispersion at the surface, and decreasing concentrations with depth is consistent with this study.

pH

The pH of seawater (7.42 - 7.92) showed little variation with depth, which is in agreement with Redvers (2000). The lowest pH value recordd is too low for seawater, and it is believed to be incorrect due to equipment freezing. This value aside, the lowest pH was sampled at site 12 (pH = 7.72), which is also the location for the waste brine from the RO plant. The pH measured in this study is in general similar, or below values recorded in previous literature. The mean pH recorded by Redvers (2000) was 7.5, however pH had an abnormally large range from 2.9 to 10.0 potentially from equipment failure. Crockett (1997) reported pH of 7.9 – 8.2 at Winter Quarters Bay, and Clark *et al.* (2009) measured an average pH of 8.1within McMurdo Sound.

Comparison was made between pH and conductivity concentrations all sites of Redvers (2000) sample run and this study. Despite the sample times being at different phases of the tide, including all available data eliminates the effect of stratification. Therefore, the mean conductivity and mean pH has remained the same.

Recommendations

It is recommended that total suspended solids be included in monitoring of the marine environment, rather than turbidity, as it gives a more accurate value of the suspended material in the seawater.

Additionally, conductivity should be included as an indicator of wastewater. The inclusion of pH is also useful to indicate if the marine environment is changing as a result of wastewater discharge.

3.4 Summary

A localised wastewater plume now exists at Scott Base, and appears to be smaller than when studied in 1999 and 1998 (Table 25). The general spatial extent of the plume is now approximately 50m long-shore, and 30m offshore, compared to Redvers (2000) when the plume was approximately 100-125m long-shore and 10-20m offshore. Many parameters have changed in the local marine environment, with no FC's now present. On average, conductivity and pH are unchanged, and DO has increased despite BOD and TOC also increasing. What is also of interest is the increase of Cu and Zn concentrations, which could be from deteriorating pipe work within the Base, or increased input from terrestrial sources.

Table 25. Comparison of the observed ranges of wastewater indicators between the present study and Redvers (2000).

Parameter	Min - Max		Mean[d]	
	Present	Redvers (2000)	Present	Redvers (2000)
FC / *E. Coli* (cfu / 100ml)	0 / 0	0 - 3550[a]	0 / 0	- [c]
Cu (µg/L)	1.2 - 26.49	0.4 - 3.2	9.86	1.1
Zn (µg/L)	12.0 - 34.92	1.0 - 4.8	3.19	2.2
NO_3^-- N (mg/L)	0.8 - 1.1	0.36 - 10.88[b]	0.5	1.18[b]
PO_4^{3-} (mg/L)	0.28 - 0.45	0.056 - 0.830[b]	0.45	0.110[b]
DO (mg/L)	10.05 - 13.02	5.2 - 11.4	12.04	9.4
BOD (mg/L)	0 - 14	1.24 -7.18	8.7	3.89
TOC (mg/L)	1.0 - 8.7	<0.3 - 2.6	2.1	0.7
Conductivity (mS/cm)	48.4 - 55.2	40.5 - 56.0	52.3	54.6
pH	7.42 - 7.92	3.9 - 10.0	7.83	7.5

[a] Faecal coliforms was the only microbial parameter analysed by Redvers (2000)
[b] Total N and P were recorded by Redvers (2000)
[c] Average of FC cannot be calculated as some data was too numerous to count
[d] For data points below the detection limit, ½ the detection limit was used

It is recommended that *Streptococus feacalis* or *Enterococci* be used as an indicator of microbiological contamination from wastewater discharge, with Cu and Zn be used as indicators of trace metal contamination. Nutrients are essential parameters for determining wastewater plumes, and it is recommended that PO_4^{3-} is primarily used, as NO_3^-- N was too often below the detection limit of the apparatus available. Dissolved oxygen should primarily be used to assess the levels of saturated oxygen in the marine environment, with BOD and TOC used for a more thorough investigation of the wastewater plume when required. As total suspended solids is a simple method to determine particulate matter in the marine environment, and should be favoured over using turbidity. As pH and conductivity have not varied greatly since 1998/1999, and do not show the plume position greatly, they may not be useful parameters to include in future monitoring.

4 Trace Element Contamination in Terrestrial Soil

4.1 Introduction

Soils are affected by the nature of the bedrock, time, climate, and ecological factors that control weathering, and the development of biotic communities. In Antarctica, chemical weathering processes are slow, and rock debris are mainly as a consequence of glacial scouring, cryoclastic processes during freeze-melt cycles, and strong winds. The ground surface is characterised by unconsolidated boulders and pebbly gravels, with fine material blown away to accumulate in the lee of larger boulders. Soils are dry and coarse in texture. In general, silt and clay contents are both less than 10% each (Bargagli, 2000). A large accumulation of water-soluble salts and very small amount of organic matter exists, with coastal areas receiving Na and Cl from marine aerosols. In coastal areas, all horizons in the active layer have an average moisture content of approximately 5%, and moisture is mainly received from melting snow and moves through the soil by capillary flow. Thin films of highly concentrated salt solutions may remain unfrozen despite the extremely low temperatures. Liquid water films with high ionic strength cause dissolution and alteration of minerals, and also the slow movement of water and ions in the active layer of soils (Bargagli, 2000).

Soil contamination in Antarctica can cause severe environmental problems, as the capability of soils to recover from contamination is limited (Allen *et al.*, 1995). The terrestrial environment at Scott Base has been significantly altered over the past 50 years by the routine operation of Scott Base. Previous studies at Scott Base (Luker, 2009; Sheppard *et al.*, 2000) indicate that elevated concentrations of metals in the soil are a result of fuel spills and the burial of metal scraps. Heavy metal contamination in soils is toxic to flora and fauna (Allen *et al.*, 1995), and as Antarctic soils are devoid of organic matter, with only sparse areas of lichens (Malandrino *et al.*, 2009), the consequences of contamination could be significant.

Soils surrounding Scott Base were investigated as a potential source of trace metals in the marine environment. Sample sites were located in areas that have a high risk of contamination, and areas that may influence the marine environment, such as along the shore adjacent to the RO intake and wastewater outfall (Figure 30). In addition, some sites were located where previous studies have been conducted (Sheppard *et al.*, 2000). This comparison will assess changes in trace elements in soils over a similar period of time to that of the marine water monitoring in the previous chapter.

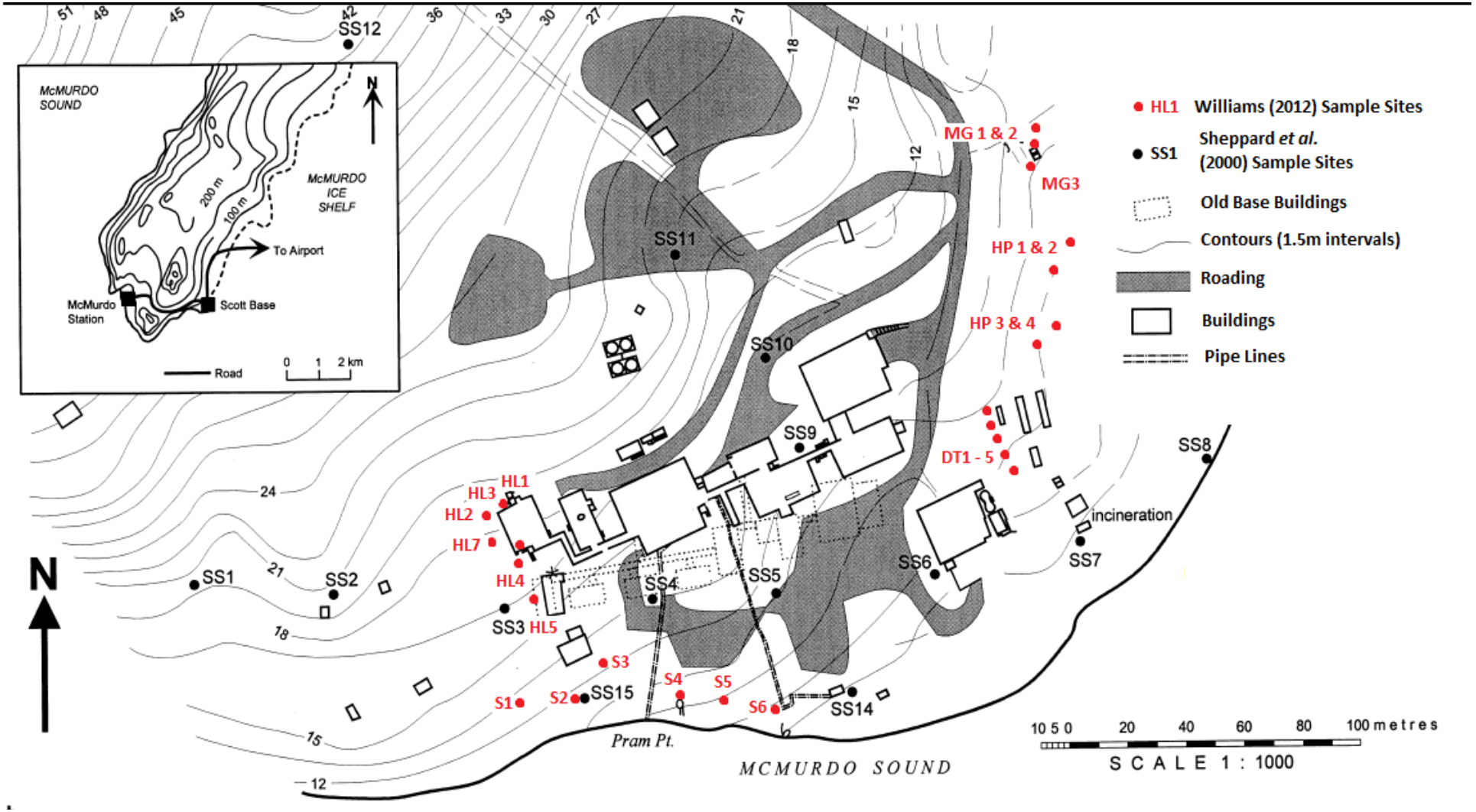


Figure 30. Soil sampling sites at Scott Base for the present study (indicated in red) and sites investigated by Sheppard *et al.* (2000) (indicated in black).

4.2 Results

4.2.1 Trace Elements in Terrestrial Soils

Soils surrounding Scott Base were analysed for a suite of trace elements (Figure 31, 32 and 33) and summarised in Table 26. In general, trace elements were elevated along the shoreline transect, and in soils surrounding the Hatherton Lab. Site S6 along the shoreline had the greatest concentrations of Cr (27.4 µg/g), Cd (0.6 µg/g), Fe (31804.4 µg/g), Mn (755.7 µg/g) and Ni (79.4 µg/g), while Pb (107.0 µg/g) was greatest at site HL5, south of the Hatherton Lab.

Copper and Zn will be the focus of this section, as they have been outlined as potential elements to monitor in the marine environment (Figure 31). The greatest Cu concentration was 66.9 µg/g in a sample collected on the north side of the Hatherton Lab, and the greatest concentration collected along the shoreline was 54.5 µg/g at site S6. At all other sites, Cu ranged from 38.8 µg/g to 11.0 µg/g. Similarly, Zn was greatest at site S6 with a concentration of 235.5 µg/g, and at the Hatherton Lab site HL6, recording the next greatest value of 157.0 µg/g. The remaining sites ranged from 17.5 µg/g to 94.4 µg/g.

Table 26. Summary of maximum, minimum and mean trace element concentrations in soil from Scott Base.

	Trace Element Concentration (µg/g)								
	As	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn
Max	13.1	0.7	27.4	67.6	31804	756	79.4	102	236
Min	1.1	0.1	5.0	11.2	7473	225	14.5	3	18
Mean	3.0	0.2	12.5	26.5	17743	453	45.4	20	65

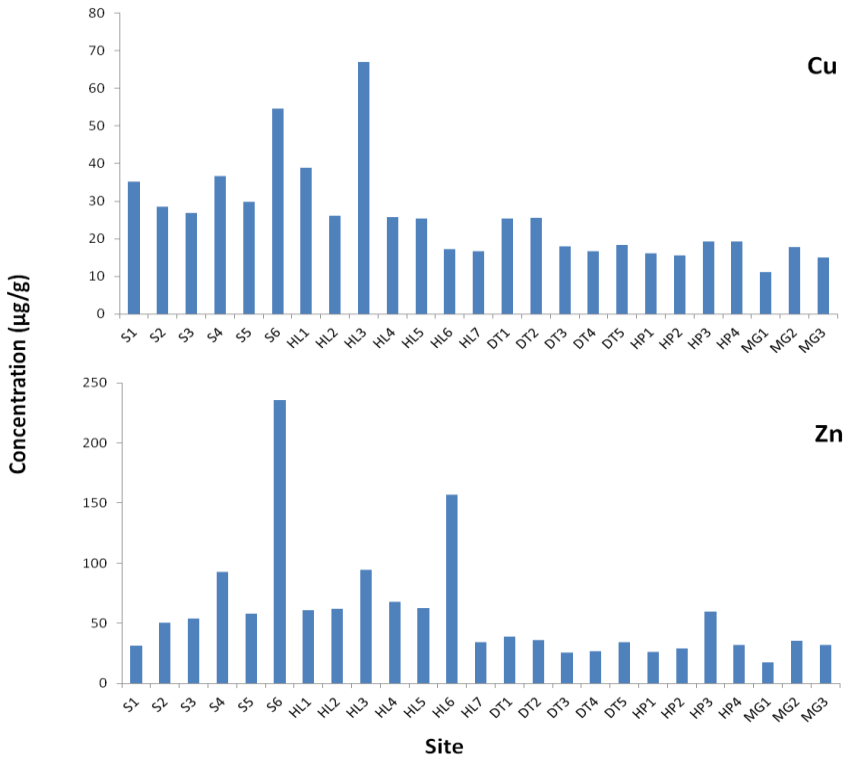


Figure 31. Concentration of total recoverable Cu and Zn in soil samples collected from Scott Base, Antarctica.

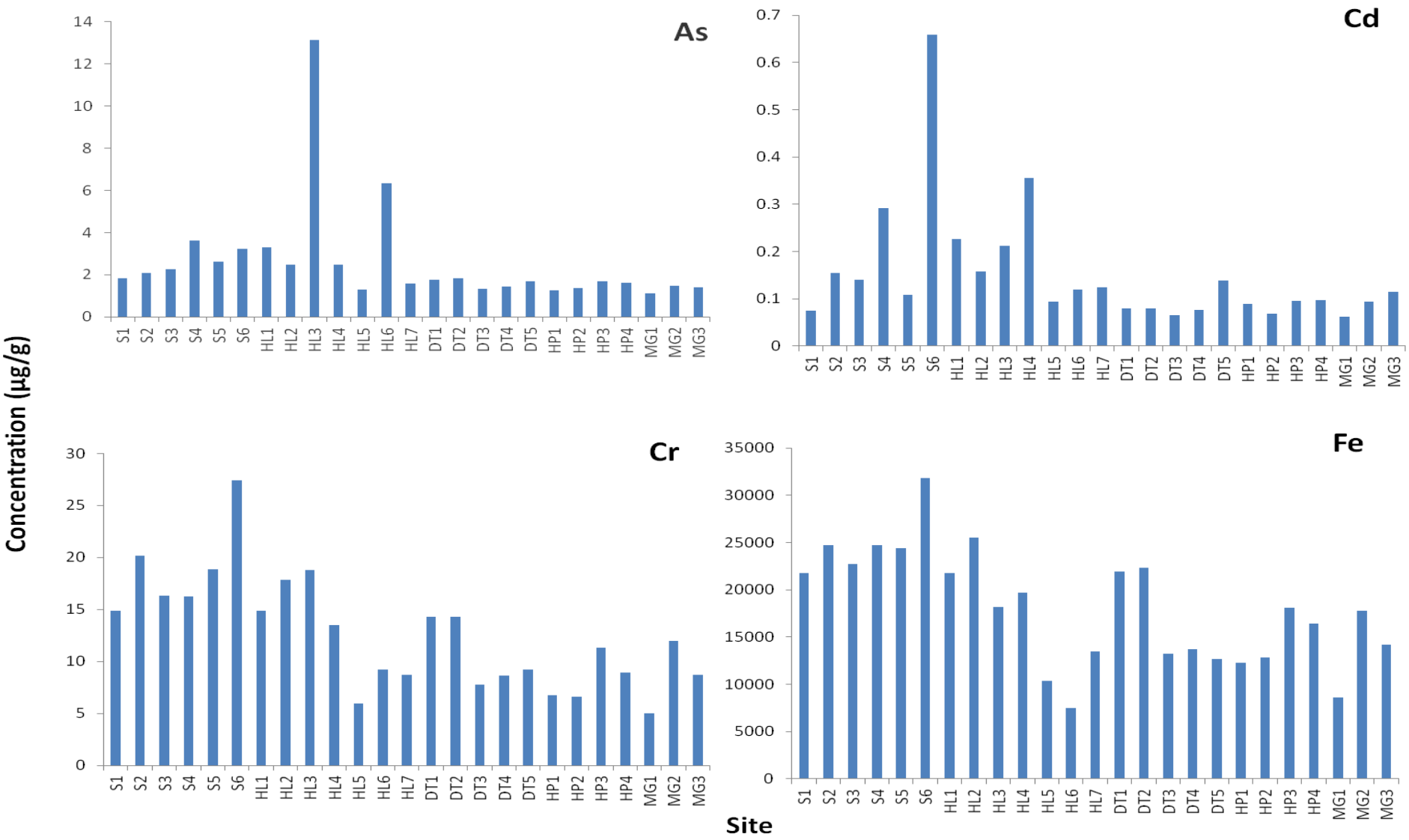


Figure 32. Concentration of total recoverable As, Cd, Cr and Fe in soil samples collected from Scott Base, Antarctica.

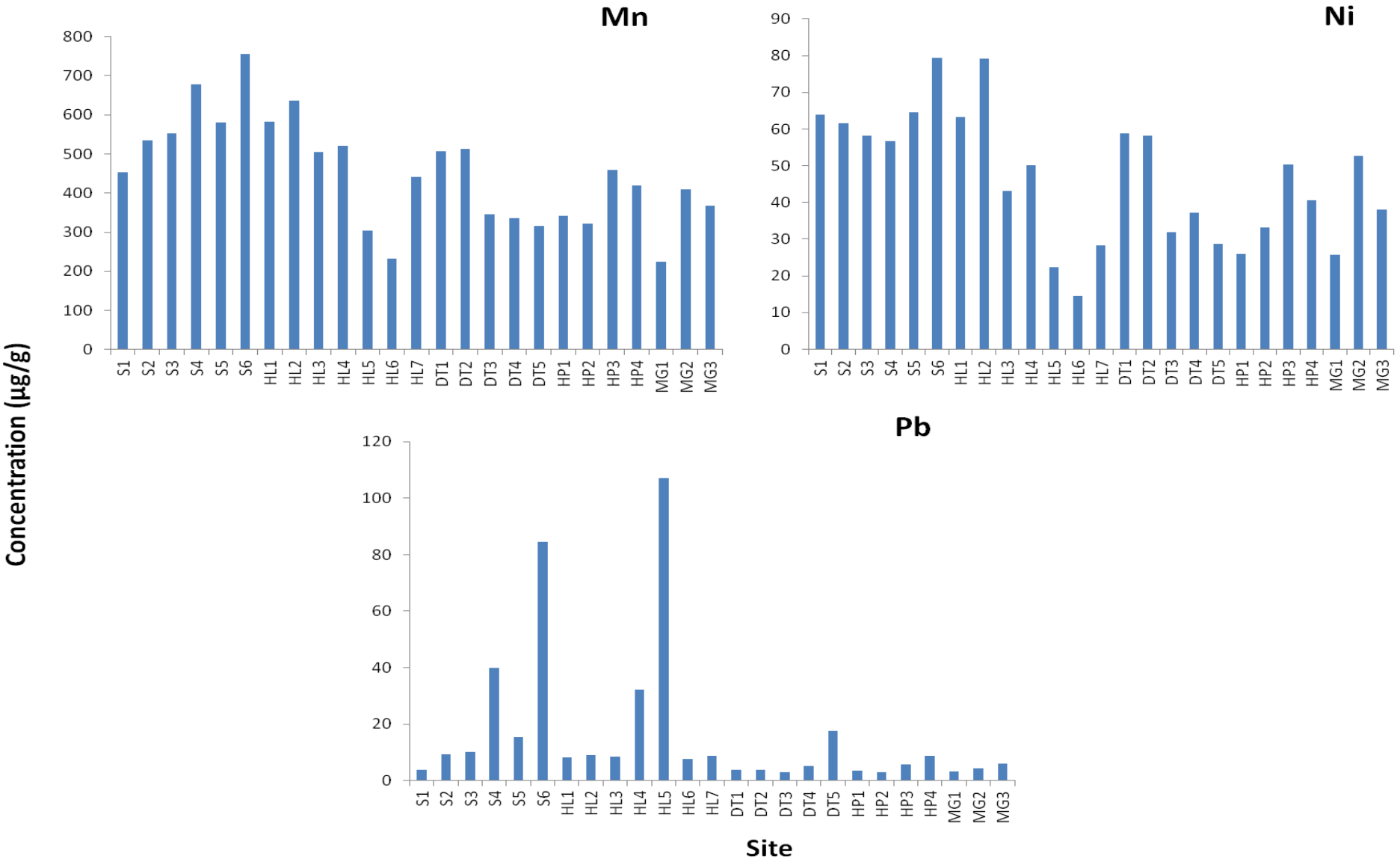


Figure 33. Concentration of total recoverable Mn, Ni, and Pb in soil samples collected from Scott Base, Antarctica.

4.2.2 Leachable Trace Elements from Terrestrial Soils

Soils were also investigated for trace elements leached with milli-q water to simulate snow and ice melt in the summer (Figure 34 and 35). The greatest leachable concentrations of As (2.53 µg/g), Fe (64.9 µg/g), Mn (3.33 µg/g), and Ni (0.09 µg/g) from the sediment were found along the shoreline around the Hatherton Lab. The greatest leachable concentration of Cu (0.72 µg/g) and Zn (1.15 µg/g) from the sediment were in a samples collected at site S6 along the shoreline. The only trace element that had the greatest leachable concentration at another site was Cr (8.5 µg/g), at the helicopter pad. Leachable concentrations of Cd and Pb in soil were below the detection limit for all sites.

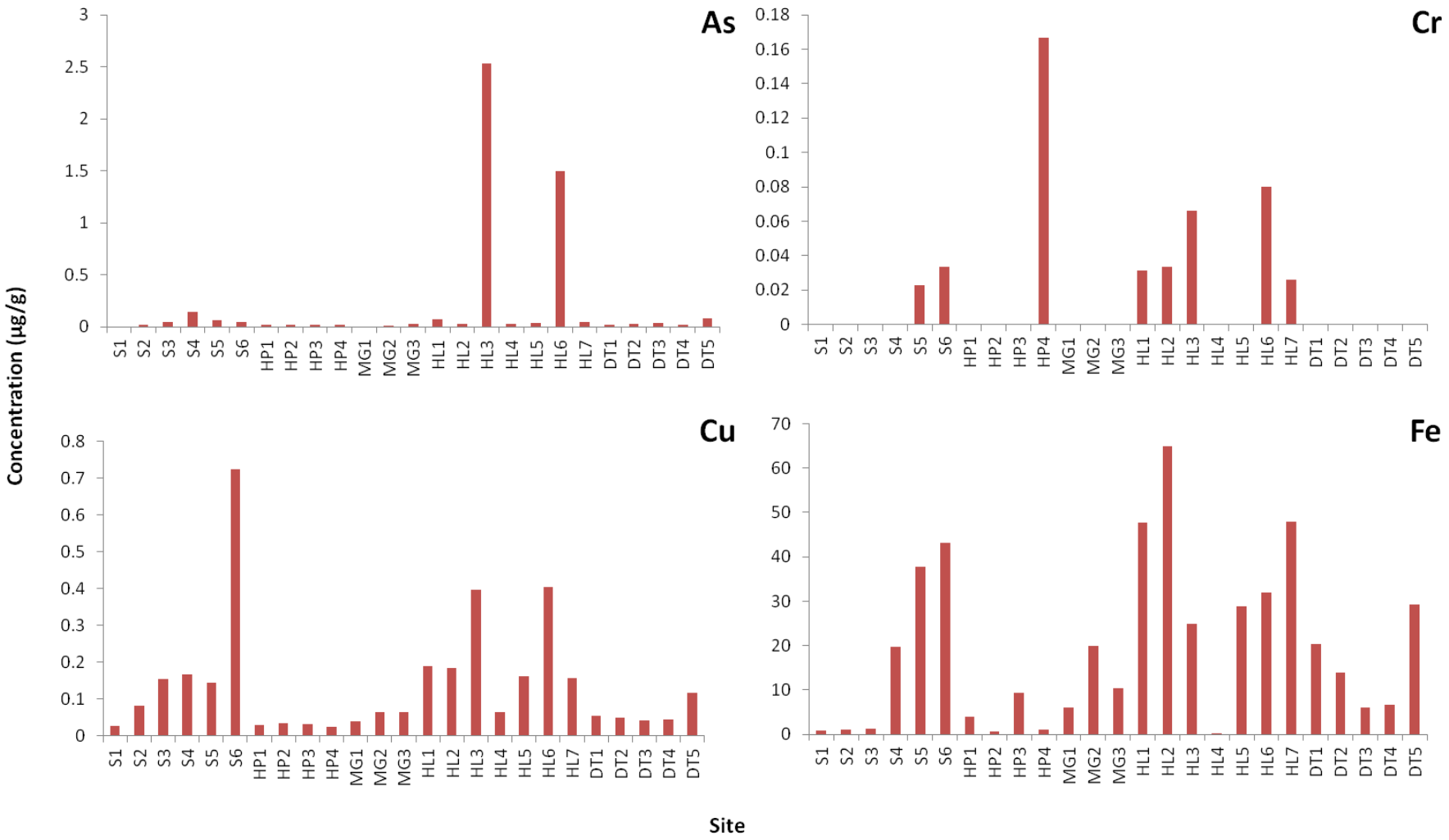


Figure 34. Concentration of water leachable As, Cr, Cu and Fe in soils samples collected from Scott Base.

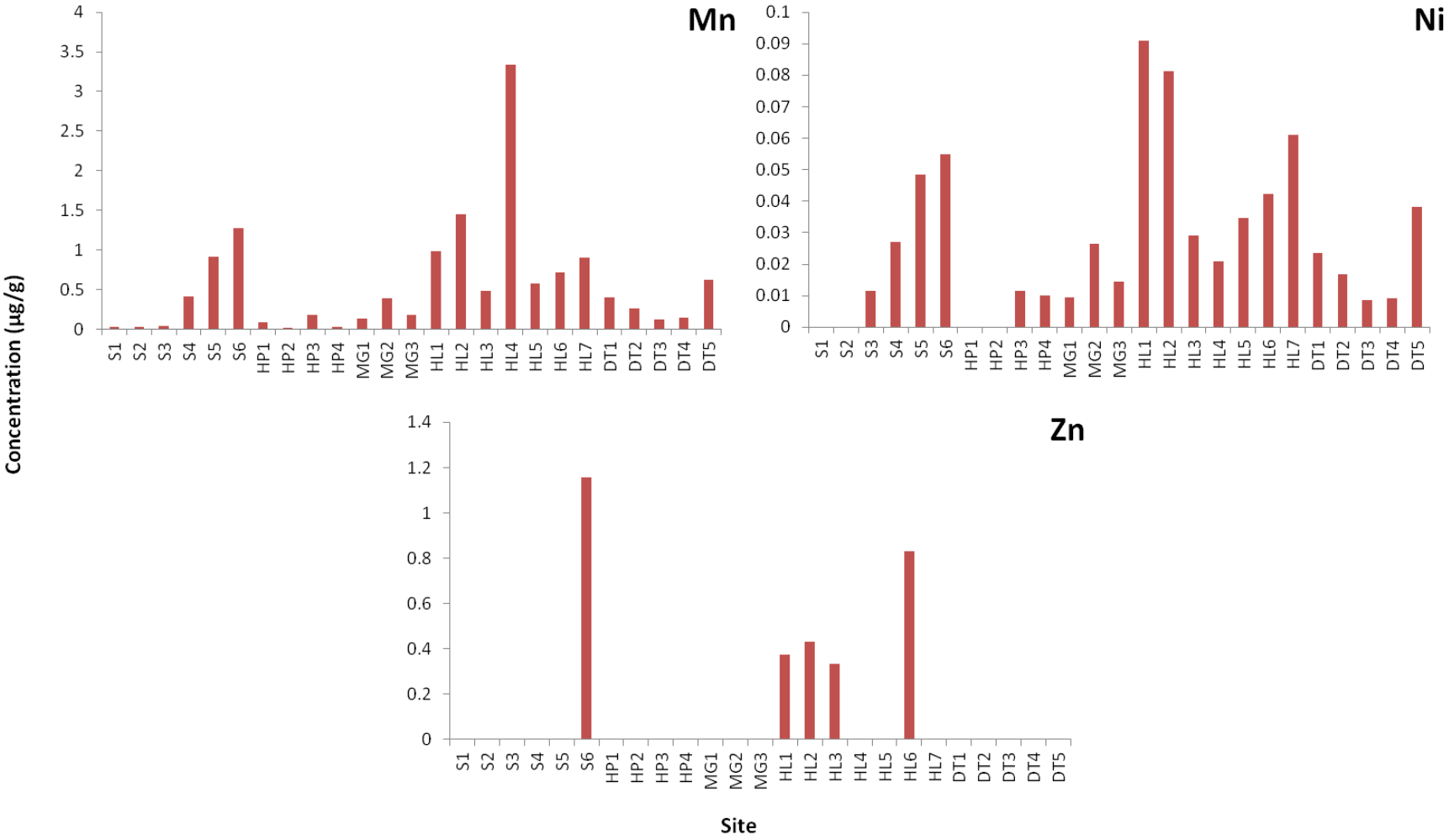


Figure 35. Concentration of water leachable Ni and Pb in soils samples collected from Scott Base.

4.3 Discussion

This study has shown that there were detectable concentrations of As, Cu, Fe, Mn, Pb and Zn in the soils surrounding the MoGas refuelling tank, the diesel tank, and helicopter pad, but in particular near the Hatherton Laboratory and shoreline transect. This is noteworthy as the sites that have a greater chance of contamination, recorded lower concentrations of trace metals, whilst the sites that are reasonably undisturbed adjacent to the Hatherton Laboratory and shore have higher concentrations. This is of interest, as those sites closest to the shore will potentially influence the marine environment, especially if transportation of trace metals is possible. To investigate transportation of trace metals in soil, snow melt was simulated with leaching experiments. Results confirm that those sites along the shore and several from the Hatherton Laboratory have higher water concentrations of leachable metals, therefore an increased probability of the trace metals being transported.

Sheppard *et al.* (2000) also investigated metal contamination in soils surrounding Scott Base (Table 27) and concluded that there was contamination by As, Cu, Pb and Zn (as noted above), and also of Ag and Cd. The study conducted by Sheppard *et al.* (2000) concluded that contamination appeared to be localised at sites where materials had historically been dumped or stored. Additionally, Sheppard *et al.* (2000) found that transport of metals was due to surface water flow, percolation through the soils and wind distribution. Claridge *et al.* (1995) also demonstrated this at Marble Point and Scott Base, with elevated concentrations of Pb, Zn and Cu in soil profiles polluted by crushed batteries, scattered rubbish and buildings. Additionally, Claridge *et al.* (1995) found that in 25-32 years, Pb had moved approximately 50cm vertically from the source.

Table 27. Comparison of water leachable trace metals from soil surrounding Scott Base (µg/L). Note that direct comparison of concentrations is not possible due to varying methods, but it can indicate what elements are elevated. Sites used for comparison were in close proximity to each other at the same depth in the soil profile.

Author	Site	As	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn
Sheppard *et al., 2000*	SS3 - A	0.015	0.0034	0.041	0.275	0.057	1	0.056	0.098	0.888
Current Study	S2	0.02	< DL	< DL	0.08	1.13	0.03	< DL	< DL	0.13
Sheppard *et al., 2000*	SS15 - A	0.028	0.037	0.07	0.225	0.095	0.0018	0.115	0.08	0.95
Current Study	HL5	0.04	< DL	< DL	0.2	28.9	0.6	< DL	< DL	0.09

4.3.1 Sources of Trace Metals in Antarctic Terrestrial Soil

Natural sources of trace metals surrounding Scott Base include basaltic volcanic rocks that dominate the Ross Island substrate. Kennicutt II *et al.* (1995) found that enrichment of Ni, Fe, Cu and Cr, and depletion of Pb and Cd, occurs in marine sediments adjacent to Ross Island. However, anthropogenic sources, such as abandoned dump sites, have been found to have far greater metal concentrations in soil. Kennicutt II *et al.* (1995) illustrated this at Palmer, and Old Palmer Stations (Cd, 22 µg/g; Cu 2100 µg/g; Pb 1600 µg/g;

Zn 3600 µg/g). Negri *et al.* (2006) compared concentrations of total hydrocarbons (THC), polychlorinated biphenyls (PCB), polyaromatic hydrocarbons (PAH) and trace metals Cu, Zn, Cd, Pb, As and mercury (Hg) in marine sediments at both Scott Base and McMurdo with less impacted sites. It was found that the mean THC concentration in sediment from Scott Base was three times greater than the pristine site, but ten times lower that samples from McMurdo Station. McMurdo station also recorded the highest concentrations of PAHs, PCBs and Cu, Zn, Pb, Cd and Hg in sediments.

At Scott Base, sites that have the greatest concentrations of trace metals are those that appear to be undisturbed at present. However, there could have been historical disturbance of some description. The Trans-Antarctic Expedition Base was closer to the shore than at present (Figure 30), but still not exactly situated at these peak sites. The high trace element concentrations recorded at sites south of the Hatherton Laboratory and shoreline are likely to be contaminated by laboratory chemicals (Sheppard *et al.*, 2000). Sites HL4 and HL5 are in the vicinity of an old storage site for equipment, and site S2 is next to the location of a building which contained a dark room for photographic processing. The high concentrations of As could arise from several sources, including airborne contamination from the incinerator, or treated wooden debris from timber being removed from existing foundations. There may also be high Pb concentrations near the helicopter pads and bases from leaded fuel as it has been observed that contamination of snow has occurred under regular flight paths due to the burning of leaded fuel, or the burning of leaded fuel in station generators and vehicles (Sheppard *et al.*, 2000; Wolff, 1990). This point may be pertinent to Scott Base, as there has also been anecdotal evidence of helicopters dumping small volumes if fuel to shut down.

Measured concentrations of trace elements from some sites at Scott Base have exceed environmental guidelines outlined by the Canadian Council of Ministers of the Environment, (1999). Numerous sites surrounding the Hatherton Laboratory exceeded Canadian Agricultural Guidelines. At site HL3, soil samples exceeded the As (12 µg/g) and Cu (63 µg/g) agricultural guidelines, HL4 exceeded the Ni (50 µg/g) guideline, and HL5 exceeded the Pb (70 µg/g) guideline. One site along the shoreline transect exceeded the Zn (200 µg/g) guideline, with all shoreline sites, and many Hatherton and Diesel Tank sites exceeding the Ni (50 µg/g) guidelines.

4.3.2 Potential Trace Element Mobility into the Marine Environment

The concern for the marine environment is possible metals leaching into the marine environment. This study found that trace elements were consistently elevated in areas adjacent to the shore (Hatherton Lab and shoreline transect), and that leachates from these sites contained greater concentrations of trace elements. The greatest leachable concentrations of soil Cu and Zn were in a samples collected along the shoreline, while As, Fe and Mn were greatest around the Hatherton Lab.

Previous studies of soils surrounding Scott Base have also indicated that trace elements are mobile in the soil, and may have the potential to reach the marine environment. Sheppard *et al.* (2000) and Anderson

and Chague-Goff (1996) indicated that Ag is mobile in soils surrounding Scott Base, and that other trace elements may have reached the marine environment. Luker (2009) also concluded that Cu and Zn can be mobile in soils around Scott Base, despite the aridity and freezing temperatures of the soil limiting the rate of movement. Studies at McMurdo Station reported high trace metals in soils, yet there was little evidence of metal contamination in subtidal sediments collected near the stations (Kennicutt II *et al.*, 1995). This finding was consistent with Claridge *et al.* (1995) who concluded that despite Zn being mobile and Cu remaining close to the source in soils of southern Victoria Land, there was no significant metal transportation of leachable metals into the marine environment. The conclusions from all the studies indicate that trace metals are present in terrestrial environment, and the soil characteristics and history of the site influence trace element mobility and subsequent effect on the marine environment.

The mobility of trace elements into the marine environment is influenced by the creation of drainage channels, the rate of thaw and melt water runoff (Sheppard *et al.*, 2000). As permafrost thaws, melt water permeates through the soil and passes through contaminated sites. The elevated concentrations of leachable trace elements found in along the shoreline transect and Hatherton Lab in the current study is probably due to the sites being more favourable mobility. A narrow channel runs underneath the Hatherton Lab down toward to marine environment, while the shoreline consists of smaller, more defined channels leading into the sea. If trace metal contamination is great enough at contaminated sites, metals will be leached into the watercourse and potentially transported into the marine environment (Luker, 2009; Sheppard *et al.*, 2000). This was proven by Goldsworthy *et al.* (2003) who conducted weak extraction techniques to determine the concentration of metals that could be leached from soil and consequently become bioavailable. Goldsworthy *et al.* (2003) noted that Cu concentrations of 2.0 mg/kg and Zn concentrations of 56 mg/kg in the soil meant that the metals could persist in forms that could be released into local water bodies with seasonal snow melt

Despite slow transportation rates of some trace metals from soils in Antarctica, environmental monitoring must be continued into the future, especially as climate change is recognised as a process that may alter soil leaching and drainage processes (Bargagli, 2000). This will ensure there is a comprehensive understanding of potential metal release rates, as other factors such as climate change evolve. This is critical because if rates increase, there needs to be processes in place to remediate polluted soil before metals enter the marine environment.

4.3.3 Remediation of Impacted Sites

It is beyond the scope of this project to provide detailed information on how to remediate contaminated sites at Scott Base. However, it is important to note that adhering to international environmental protocols can minimise the impact of trace elements in soils, despite anthropogenic disturbance on the terrestrial environment at Scott Base being largely unavoidable (Luker, 2009). Prevention is the easiest option for protecting the Antarctic environment, yet for those sites that have been highlighted as potential sources of contamination, future studies must assess to what degree remediation can be implemented. Several

factors must be considered when investigating remediation of impacted sites, including the type of heavy metal present, and the local environment (Luker, 2009). Tin *et al.* (2008) also highlighted the need for research to focus on developing low-cost methods of remediating contaminated sites as logistical difficulties and environmental risks make large scale excavation difficult. Methods such as bioremediation have already been considered by Argentina, Australia, France, USA, UK and New Zealand, and despite the methods taking longer, the end result if controlled will be low cost (Tin *et al.*, 2008).

When comparing data to standards and other sites within Antarctica, caution must be exercised. Present standards and trigger values for protecting the ecology of Antarctic soil are derived from temperate organisms, thus the values may be overly conservative for the protection of Antarctic ecosystems (Snape *et al.*, 2008). In future research, standards should be created for areas surrounding research stations. The need for site specific standards is due to differences in mineralogy, grain size and organic matter between sites and samples, and this will ultimately affect the content of trace metals in samples. However, the use of standard dissolution and extraction procedures is also important, and the methods used in this study followed prescribed methodology to enable comparison with previous studies, especially from Scott Base.

4.4 Summary

Soil digests from Scott Base showed areas adjacent to the shore that had greater concentrations of trace elements than sites further inland. This may mean a higher risk of contamination to the marine environment as trace elements have also been found to be partly water leachable at these same sites. Despite mobility of trace elements from soil being slow in Antarctica, soils at Scott Base have characteristics that promote trace element mobility, and may have the ability to impact on the marine environment into the future.

5 Characterisation of Scott Base Domestic and Drinking Water

5.1 Introduction

It is of paramount importance that safe water is provided to communities for domestic purposes, including drinking. This is even more important for remote outposts such as Scott Base. Due to the proximity of the wastewater outfall to the RO intake at Scott Base, there is a possibility that wastewater contaminants could enter into the reverse osmosis plant. This study investigated whether FC/*E. coli*, trace metals, and nutrients from the wastewater discharge were detected at the reverse osmosis intake, or in the RO water in the Base. Total suspended solids and pH were also measured in RO1 and RO2 water to determine if the wastewater discharge altered the aesthetic characteristics of the freshwater on base.

The RO water samples were compared to maximum acceptable values (MAV's) outlined in the Drinking-water Standards for New Zealand (Ministry of Health, 2008) to investigate compliancy. MAV's are included for microbial determinands, such as *E. coli*, and chemical determinands such as trace metals and nutrients. Other determinands are given as guidelines values, such as pH and total dissolved solids.

5.2 Results

5.2.1 Faecal Coliforms / *Escherichia coli*

Faecal coliform or *E. coli* were not detected in all samples collected at the RO intake, or RO2 (drinking) and RO1 (domestic) water.

5.2.2 Trace Metals

Domestic and drinking water samples collected at Scott Base were analysed for the same elements as within the plume (Figure 36). For comparison, data is plotted against seawater at site 12 (at the RO intake) and at the pump house.

The results of this study indicate that Cu, Ni and Zn are introduced at the pump house (Figure 36). As seawater is processed through the RO plant to become fresh drinking water, Zn and Ni are significantly reduced. However, Cu concentrations increases during RO to be highest in drinking water than. The total Cu concentration of the RO2 water was 98.8 µg/L, and the dissolved concentration was 92.7 µg/L.

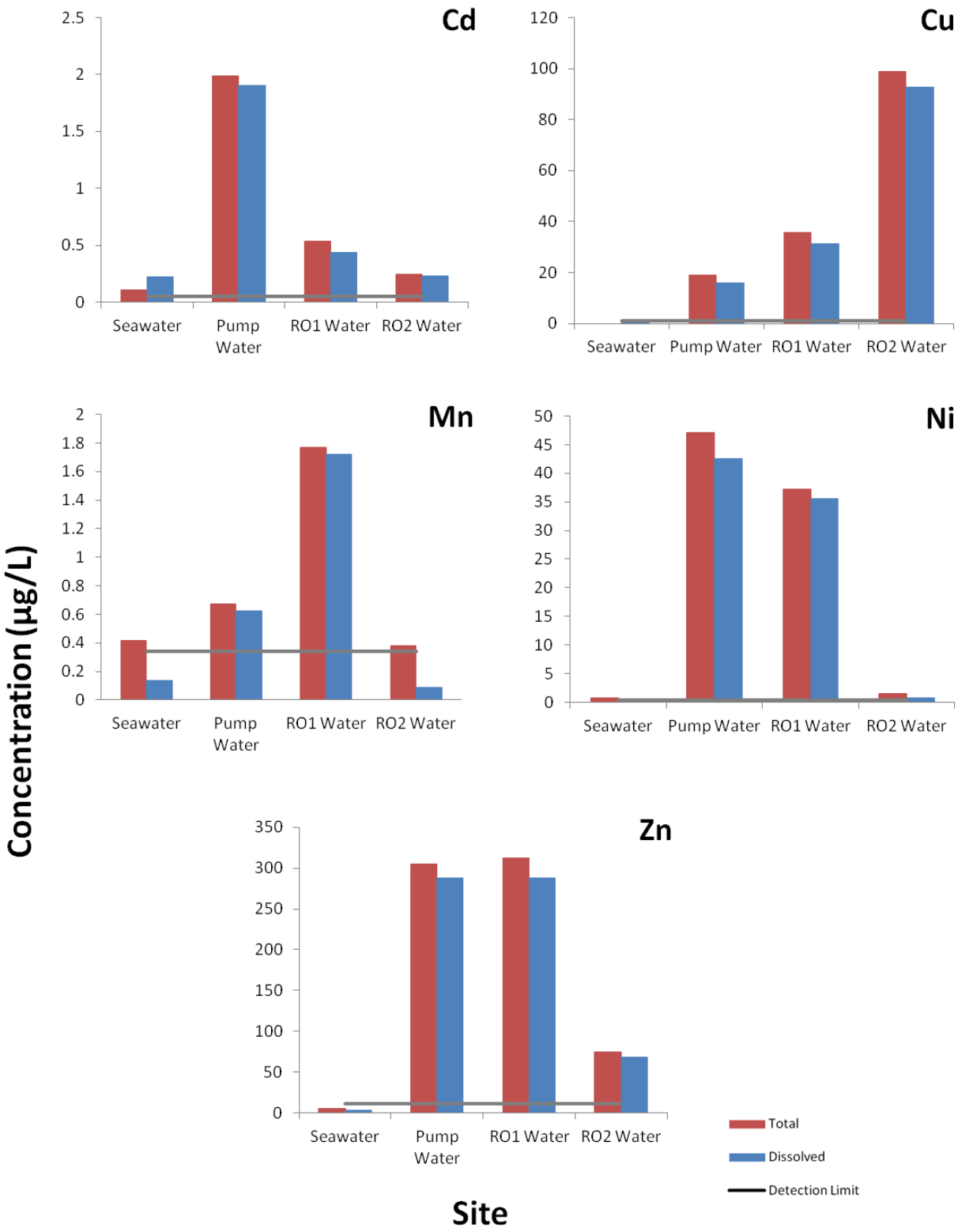


Figure 36. Trace metal concentrations of RO1 and RO2 water, with seawater samples collected from site 12 (adjacent to the intake) and at the pump house for comparison.

5.2.3 Nutrients

Concentrations of NO_3^-- N were less than the detection limit on the 9/11/2010 and 17/11/2010. The RO process significantly reduces the concentrations of PO_4^{3-} in RO1 and RO2 water (Figure 37). However, on sample run one PO_4^{3-}concentrations increased by 0.15 mg/L, and on sample run two, NO_3^-- N increased by 0.1 mg/L between the RO1 and drinking water samples.

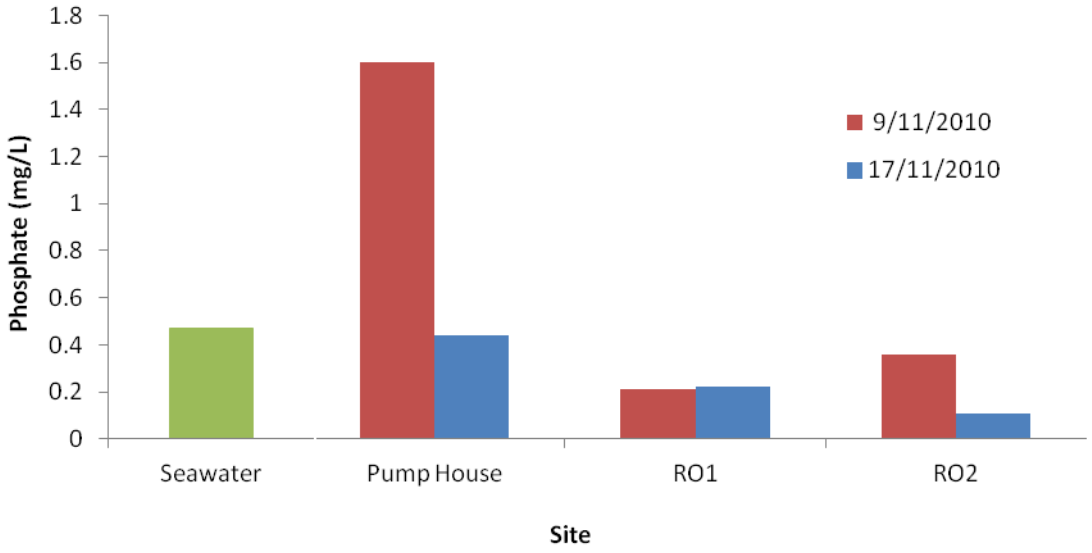


Figure 37. Phosphate concentrations of RO1 and RO2 samples, with seawater samples collected site 12 (indicated in green) and at the pump house for comparison.

5.2.4 pH and Total Suspended Solids

The pH increases during the first RO process, yet is reduced again in RO2 water (Figure 38). Total suspended solid concentrations were not present in RO1 and RO2 samples.

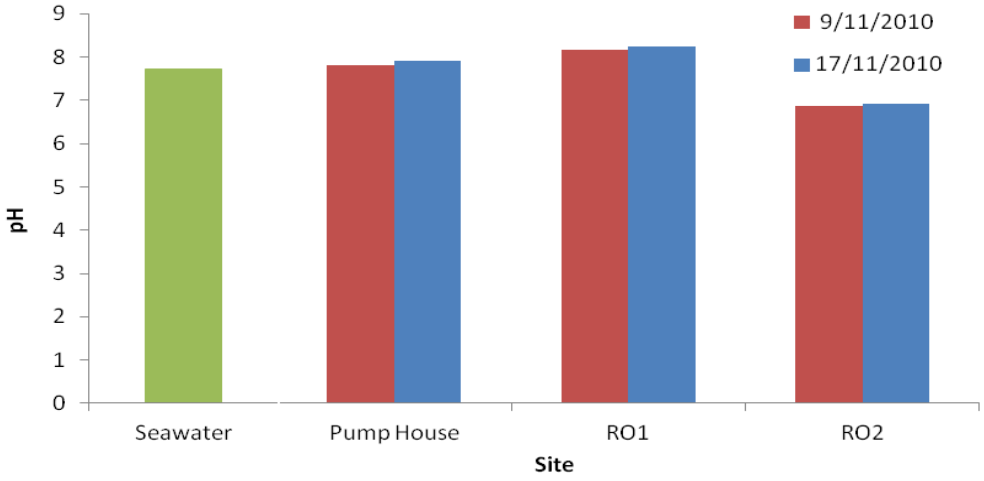


Figure 38. pH of RO1 and RO2 samples, with seawater samples collected site 12 (indicated in green) and at the pump house for comparison.

5.3 Discussion

Protecting the health of those people at Scott Base is of paramount importance, as the risk of spreading illness is high in confined living quarters, and advanced medical assistance is limited. The RO water samples in this section are used to investigate whether the wastewater plume reaches the RO intake, and also compared to Drinking-water Standards for New Zealand (Ministry of Health, 2008).

As the Scott Base community lives and work within confined quarters, the risk of contaminated drinking water is higher than in normal domestic settings. This risk is increased even greater as there is only one source of drinking water. The most probable effects of contaminated drinking water are gastrointestinal infections, such as campylobacteriosis and diarrhea, caused by bacterial, viral or parasitic organisms entering the drinking water (Ministry of Health, 2008).

5.3.1 Faecal Coliforms / *Escherichia coli*

Faecal coliform or *E. coli* were not detected in the domestic (RO1) or drinking water (RO2) at Scott Base. This result was below the MAV for *E. coli* which is less than one in 100 ml of sample. As FC or *E. coli* were not detected in the marine environment, this parameter cannot be used to investigate if the plume is reaching the RO intake. However, the water entering the RO intake has been monitored for FC on a more regular basis by Antarctica New Zealand. Results confirmed that the plume at times, extended to the RO intake which is 75m from the outfall. The FC concentrations ranged from 0-5550 cfu/100ml (median 55 cfu/100ml, for n=119) (Antarctica New Zealand, 2001). Also, as part of Antarctica New Zealand's routine monitoring, RO2 water is analysed for FC, and congruent to this study, no FC were observed in the previous three years.

5.3.2 Trace Metals

Analysis of water samples for trace metals indicate that Zn, Ni, Fe and Cu are not being introduced from wastewater discharge, or runoff from the terrestrial environment, but in the pump house. However, as the water is processed through the RO plant, Zn and Ni are significantly reduced. Cu increases during the RO process to be greater in drinking water than in any other water samples collected in this study.

Trace metals in drinking water can cause taste and staining problems. Fe, Cu, Mn and Zn arising from pipes and fittings in the distribution system are the most common elements to cause such problems. Elevated Cu concentrations have been found to be due to corrosion of plumbing (Ministry of Health, 2008). The metallic taste of Cu in drinking water has been found to cause nausea in humans, and elevated Cu concentrations can cause liver failure in susceptible individuals (Stern, 2010). Zn by comparison is an essential requirement for humans, yet toxic levels inhibit the absorption of Cu and Fe.

The RO2 at Scott Base meets the maximum acceptable values (MAV) for trace metals in drinking water as specified in the Drinking Water Standards for New Zealand (Ministry of Health, 2008) (Table 28). Cu is also under the guideline values (GV) for aesthetic determinands of 1000µg/L. Despite trace metals

being under the GV and MAV's for drinking water, it may be useful in the future to monitor concentrations to assess the effects of the base's plumbing and RO plant. This would require a more thorough investigation of the reticulated freshwater at Scott Base. Sampling immediately prior and after reverse osmosis, and at points throughout the base, would indicate where Cu is being introduced and if it is the RO plant or pipe work that is causing the contamination.

Table 28. Ministry of Health (2008) MAVs compared to trace metal concentrations for two RO2 water samples from Scott Base.

Element	MAV (µg/L)	Total Metals (µg/L)		Dissolved Metals (µg/L)	
		9/11/2010	17/11/2010	9/11/2010	17/11/2010
As	10	0.14	0.21	0.09	0.1
Cd	4	0.24	0.25	0.24	0.22
Cu	2000	96.5	101	94.2	91.2
Pb	10	1.46	1.55	1.01	1.35
Mn*	400	<1	<1	<1	<1
Ni	80	1.62	1.3	0.92	0.7
Zn	1500	74.8	75.8	77.2	60.9

*Mn is a GV only and has no MAV for drinking water.

The Drinking-Water Standards for New Zealand (Ministry of Health, 2008) are based on the consumption of 2 litres of water consumed daily. However, the recommended daily drinking water intake for persons in Antarctica is much greater. Despite Scott Base meeting the MAV's and GV's for drinking water, the greater amount of water consumed by its occupants may mean the MAV's and GV's are under estimated for human health.

5.3.3 Nutrients

All samples analysed for NO_3^-- N were below the detection limit, and cannot be compared to existing MAVs or used to indicate if the plume extends to the RO intake.

Phosphate concentrations at the pump house are similar to average concentrations in the marine environment, indicating that the plume may extent to the RO intake. As increased concentrations of phosphate are not a significant health issue, and there is no MAV, there is little concern from this parameter. However it is useful to note that the RO treatment does reduce concentrations of PO_4^{3-}. Samples collected on the 9/11/2010 had elevated concentrations of PO_4^{3-} at the pump house, and concentrations were reduced again in the drinking water and to domestic water.

5.3.4 pH and TSS

The pH of water primarily affects the taste of water, with a high pH having a soapy taste and feel (Ministry of Health, 2008). The reduced pH in RO2 is due to the removal of ions buffering the pH to approximately 8.0 in seawater. Samples tested for pH (6.86 – 6.92) were below the guideline value for aesthetic determinands (pH = 7.0 – 8.5), but an acceptable level for drinking water (Table 29). With

comparison to the McMurdo drinking water (Lisle *et al.*, 2004), pH (Table 30) shows that this is above that of Scott Base and is slightly above the GV for New Zealand.

Table 29. pH concentrations compared with MAV.

Determinand	GV	9/11/2010	17/11/2010
pH	7.0 - 8.5	6.86	6.92

Table 30. McMurdo Station drinking water pH (Lisle *et al.*, 2004).

Sample Site	pH
Treatment Plant	8.71 ± 0.42
Dormitory	8.71 ± 0.27
Vehicle Shop	8.65 ± 0.21

5.4 Summary

Concentrations of FC / *E. coli*, trace metals, and nutrients were below the MAVs stated in the New Zealand Ministry of Health's Drinking Water Standards. However, elevated concentrations on Cu and Zn were observed in RO2 water, and analysis of these parameters may be useful in the future. Monitoring of drinking water could not only be used to maintain human health quality, but could be used to monitor the state of copper piping that delivers the water around the base. pH was outside the guideline values stated in the New Zealand Ministry of Health's Drinking Water Standards, but is unlikely to have any significant health effects.

6 Recommendations and Synthesis

This chapter discusses the key results, and brings together data from this study to deliver recommendations for future study, including how Antarctica New Zealand can reach monitoring goals in an effective and efficient manner, and fulfil obligations under the Environmental Protocol.

To remain at the forefront of environmental protection in Antarctica, conservation requires the collective development of standardised monitoring that produces robust and readily available data (Bargagli, 2000). This study has shown that there are valid options available to conduct sound monitoring in Antarctica, and what is now required is motivation from national Antarctic programs to carry out such research.

This study enables Antarctica New Zealand to fulfil mandatory obligations under the Environment Protocol to understand the environmental impacts of Scott Base. Information provided outlines how improved wastewater treatment at Scott Base has reduced impact on the Pram Point marine environment, and how soil contamination has the potential to enter the marine environment. Additionally, it has provided baseline information on the quality of the reticulated freshwater on Base, and the compliancy with New Zealand Ministry of Health drinking water standards.

6.1 Key Findings

- **Characterisation and Extent of the Wastewater Discharge from Scott Base**

The current treatment practices are effectively reducing concentrations of contaminants, as indicated by the pre- and post-treatment data. The increased quality of the wastewater discharge is reflected in the marine environment, with concentrations of typical indicators (Table 31) frequently below detectable levels.

Table 31. Minimum and maximum concentrations of wastewater indicators offshore from Pram Point.

Parameter	Min - Max Concentrations
FC / *E. Coli* (cfu / 100ml)	0 / 0
Cu (µg/L)	1.2 - 26.5
Zn (µg/L)	12.0 - 34.9
NO_3^-- N (mg/L)	0.8 - 1.1
PO_4^{3-} (mg/L)	0.28 - 0.45
DO (mg/L)	10.05 - 13.02
BOD (mg/L)	0 - 14
TOC (mg/L)	1.0 - 8.7
Conductivity (mS/cm)	48.4 - 55.2
pH	7.42 - 7.92

The general spatial extent of the plume is now approximately 50m long-shore, and 30m offshore. This is thought to be due to continual, low flow discharge percolating into the marine environment. Despite the currents of Pram Point not being suitable for rapid initial dispersion (Redvers, 2000), the continual and low flow means that wastewater is well mixed, as indicated by the stratification results.

Redvers (2000) described the dispersion and fate of sewage and wastewater in the marine environment offshore from Pram Point prior to the WWTP being installed. Results of Redvers' (2000) study indicated that the wastewater plume was detected in an area approximately 150 - 175, long shore, and 40 - 50m offshore from the outfall.

Redvers (2000) data has been used as a baseline for the quality of the Pram Point marine environment prior to installation of the WWTP. The present study has shown that the installation of Antarctica New Zealand's WWTP at Scott Base has increased the quality of the wastewater discharge. As a consequence, the extent of the wastewater plume is reduced and the quality of seawater has increased in the local marine environment. Since Redvers (2000), the spatial extent of the plume has reduced by approximately 100-125m long-shore and 10-20m offshore. Many parameters have changed in the local marine environment, with no FC now present. On average, conductivity has decreased, and DO has increased despite BOD and TOC also increasing. From the limited data available for comparison, Cu and Zn have increased in the marine environment, which could be due to increased concentrations in the discharge into the marine environment.

Routine monitoring data from April 2009 to Janurary 2012 of wastewater discharge from Scott Base, and seawater from the RO intake, indicate that concentrations of FC and BOD_5, are elevated over the summer months, especially April. FC's at the WWTP range from 1000 – 874000 CFU /100ml, with counts being too numerous to count on several occasions. These high counts are recorded in April, coinciding with the end of the summer field season. At the RO intake, FC have rarely been detected. When FC have been detected, it has been during the period of December to April, and concentrations have ranged from 1 to 1000 CFU /100ml. Similarly, BOD_5 is elevated during the summer months, reaching peak concentrations of 49 mg/L in April 2010, and 26 mg/L in April 2011.

Analysis of the routine WWTP monitoring found that summer data prior to 2009 is often missing pieces of information, or missing complete months. However, in the past year, monitoring has been conducted on a regular monthly basis. Collecting data over a full year, especially in summer, is important as this is when the base is running at maximum occupancy and impacts may be greatest.

- **Scott Base soil as a potential source of trace elements in the marine environment.**

Analysis of soils surrounding Scott Base illustrate that historical disturbance has left its legacy. Higher concentrations of As, Cd, Cu, Pd and Zn were measured in areas that were previously used to store chemicals, or built on. Simulating snow melt indicated that these trace metals may be leached into the marine environment, and especially from those sites closest to the shoreline.

- **Characterisation of the Drinking Water at Scott Base**

All measured parameter concentrations were below the Ministry of Health drinking water standards for faecal coliform / *E. coli*, trace metal, nutrient, pH and TSS. However, as concentrations of Cu and Zn were observed in RO2 water, analysis of these parameters may be useful in the future. Monitoring of drinking water could not only be used to maintain human health quality, but could be used to monitor the state of copper piping that delivers the water around the base

6.2 Recommendations

Past research has highlighted that even small amounts of sewage discharged from research stations may have cumulative environmental impacts, yet few stations treat wastewater. Connor (2008) champions countries to adopt technologies that best treat wastewater and goes further to add that technologies need not be high-tech, or expensive in order to reduce impacts. With these factors in mind, there is now little excuse not to treat wastewater in Antarctica. The other significant concern is national Antarctic programmes failing to have comprehensive monitoring programmes set in place (Tin *et al.*, 2008), despite information on techniques (GERG, 2000) being readily available, and cost effective.

It is also essential to ensure that data collected is critical, scientifically defensible and can be used for understanding both ecosystem processes and for modifying human activities in Antarctica (Walton, 1998). Additionally, methods need to be standardized so that data can be effectively compared. Different methods determining the same contaminant can produce contrary results (Sawyer *et al.*, 1994), and adoption of the Antarctic Environmental Monitoring Handbook (GERG, 2000) by national Antarctic programmes would eliminate this problem.

The adoption of standard techniques, and having comparable data available would enable Antarctic specific water and wastewater standards to be created. Such standards would provide a common goal for all nations that occupy bases on the continent, rather than using national standards from far afield that are not suitable for a pristine environment such as Antarctica. Information on the response of Antarctic species to contaminants is also needed to ensure that environmental standards that are written into legislation are appropriate for the local Antarctic environment (Duquesne *et al.*, 2000).

6.2.1 Scott Base Wastewater Treatment

Despite the many challenges to wastewater treatment, Antarctica New Zealand has succeeded in treating wastewater and reducing the impact of discharge on the environment. Issues recorded at other research stations include the need for treatment plants to be operated in exclusion from living and working areas, insulated and/or heated transfer lines, and the large seasonal variation in station populations requiring alterations of treatment methods. These issues have been overcome by planning and construction of Scott Base's wastewater plant to be isolated from the main base facilities, easily accessible, and with a plant that can operate at optimal levels with a range of occupants on base. The success of Scott Base's ozone system for secondary treatment is also beneficial, as UV technology has not always been successful at removing microbial contaminants (Smith and Riddle, 2009).

6.2.2 Scott Base Monitoring Program

Monitoring the impact of Scott Base on the local environment needs to be continued into the future as it is important and mandatory obligation under The Madrid Protocol. It is essential that results are communicated to the greater Antarctic community, and for the continuity of Antarctica New Zealand's monitoring of the WWTP.

For monitoring to be sustainable, it requires clear research objectives, with economical and practical methods (Sawyer *et al.*, 1994). It has been recognised that current monitoring of the wastewater discharge from Scott Base requires a balanced program to answer both environmental and practical engineering questions. From an environmental management perspective, monitoring should provide information regarding the impact of the base on the local environment, and if current practices are minimising such impact to the desired level. From an operational and infrastructure standpoint, information is sought on whether the wastewater treatment and RO plants are working efficiently and effectively.

By answering the above environmental and engineering questions, Antarctica New Zealand is able to effectively review current monitoring, and is in a position to set the bar high for future monitoring. To effectively review monitoring, and create a program that is streamlined to other national Antarctic programmes, it is recommended that the three-step approach to environmental monitoring (Figure 3) be adopted from the working paper from ATCM XXXV (COMNAP, 2005). Adoption of the three-step approach is feasible for Antarctica New Zealand, as many aspects of the process have been completed. For example, results of this study can be used as a baseline for future monitoring of the wastewater discharge from Scott Base on the marine environment (Step 1). Additionally, sampling methods are easily defined (Step 2) by adopting the recommendations below, and updating current methods.

6.2.3 Recommended Indicators of Monitoring Contamination at Scott Base

Wastewater Contamination

Redvers (2000) concluded that the most suitable indicators of effluent contamination were total nitrogen, total phosphorus, faecal coliform and TOC. As the wastewater discharge is now treated, the suitability of wastewater parameters must be reassessed. Together with information provided in the Antarctic Environmental Monitoring Handbook (GERG, 2000), the following parameters are recommended for monitoring the wastewater plume in the Pram Point marine environment.

- Faecal coliform, and *Escherichia coli*
- Copper and Zinc
- Nitrate - Nitrogen and Phosphate
- Dissolved Oxygen, and if necessary Biological Oxygen Demand, and Total Organic Carbon
- Conductivity
- Total Suspended Solids
- pH

In some cases methods can be improved to make the monitoring program more efficient, whilst retaining integrity. For example, the use of 3M petri-films would save a significant amount of time for microbial counts, instead of the current method of preparing agar plates.

Future Study and Additional Wastewater Indicators

Monitoring of seawater should also include *Streptococus feacalis*, or *Enterococci*, chemical oxygen demand, and emerging contaminants. Emerging contaminants are chemical and microbial parameters that are derived from wastewater sources and are increasingly being recognised as a serious pollutant. Historically, emerging contaminants have not been considered detrimental to the environment, and as a consequence, wastewater treatment practices often are not designed to remove such parameters. For example, Hale *et al.* (2008) found that improved wastewater treatment practices at McMurdo Station and Scott Base were insufficient to limit the local dispersal of polybrominated diphenyl ethers (PBDE).

Bio-monitoring and bio-indicating species could also be used to measure impacts on the local environment. Organisms readily available at Pram Point, and suitable for impact assessments include phytoplankton (GERG, 2000), gammarids (Duquesne *et al.*, 2000) or benthic infauna (Anderson and Chague-Goff, 1996). However, in shallow areas of Pram Point (to depths of 30m), the steep slope and loose substrate results in few large invertebrates being present large invertebrates that are often used for bio-monitoring (Battershill, 1989).

Soil Contamination

Monitoring the mobility of trace metals in soil should use Cu and Zn, at five yearly intervals. These trace elements have been found to be the most common contaminants at Scott Base, and baseline data is available (Luker, 2009). Additional monitoring of soil should also include total petroleum hydrocarbons (Goldsworthy *et al.*, 2003), total organic carbon, and polycyclic aromatic hydrocarbons (GERG, 2000; Watier, 2008). Note that soils in this study were not analysed for the above parameters as metal transport between terrestrial and marine environments was the priority.

Drinking Water Contamination

The current use of FC for indicating drinking water contamination can be expanded to include *E. coli*, *Streptococus feacalis* or *Enterococci*, as it more resistant to environmental stress and is closely associated with human and animal faeces (Owili, 2003).

Copper and zinc should also be monitored in drinking water on two yearly intervals. Copper in particular is a suitable parameter to monitor for not only the health of the occupants of Scott Base, but also to record the fitness of the piping on base.

7 References

3M. (2008). Interpretation Guide: Petrifilm E.coli/Coliform Count Plate *3M Asia Pacific*. Singapore.

Ahammer, H., Matz, T., El Naggar, S., & Gernandt, H. (2000). *Biological sewage plants at Neumayer and the Argentinian Base Jubany Antarctica.* Paper presented at the Proceedings of the 9th SCALOP Symposium, 12 July 200, Tokyo.

Allen, H., Huang, C., Bailey, G., & Bowers, A. (1995). Kinetics of metal sorption reactions *Metal speciation and contamination*. Boca Ration: Lewis Publishers.

Anderson, B., & Chague-Goff, C. (1996). Benthic Forminifera and Trace Metals in Sedimants off the Scott Base Sewer Outfall, Antarctica *Antarctic Data Series Number 18*: Victoria University of Wellington, Research School of Earth Sciences.

Antarctica New Zealand. (1999). Environmental monitoring results in relation to the Scott Base sewage and wastewater outfall (raw data). In G. Redvers (Ed.), *Dispersion and Fate of Sewage and Wastewater Components From Scott Base, Antarctica.*

Antarctica New Zealand. (2001). Initial Environmental Evaluation (IEE) for Scott Base Wastewater Treatment. from Environmental Impact Assessment Database

Antarctica New Zealand. (2011). Environmental monitoring results in relation to the Scott Base sewage and wastewater outfall (raw data).

ANZECC. (2000). Australian and New Zealand Guidelines for Fresh and Marine Water Quality. In Australian and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand (Eds.), *National Water Quality Management Strategy* (Vol. 1).

Aronson, R. B., Thatje, S., McClintock, J. B., & Hughes, K. A. (2011). Anthropogenic impacts on marine ecosystems in Antarctica. *Annals of the New York Academy of Sciences, 1223*, 82-107.

Bargagli, R. (2000). Trace metals in Antarctica related to climate change and increasing human impact. *Reviews of Environmental Contamination and Toxicology, 166*, 129-174.

Bargagli, R. (2006) Antarctic ecosystems: environmental contamination, climate change, and human impact. *Ecological Studies, v. 175.* Berlin: Springer Verlag.

Bargagli, R. (2008). Environmental contamination in Antarctic ecosystems. *Science of the Total Environment, 400*(1-3), 212-226.

Barnes, D. K. A., & Clarke, A. (1995). Seasonality of feeding activity in Antarctic suspension feeders. *Polar Biology, 15*(5), 335-340.

Battershill, C. N. (1989). Distribution and Abundance of Benthic Marine Species at Cape Armitage, Ross Island, Antarctica - Initial Results. *New Zealand Antarctic Record, 9 (2)*, 35-52.

Battershill, C. N. (1992). The ecology of the Pram Point reef slope, Ross Island, Antarctica. *Antarctic Record, 12*, 17-18.

Biletnikoff, N., Shenk, C., Van Velduizen, D., & Knuth, M. (2006). *Wastewater Treatment in Antarctica: Challenges and Process Improvements.* Paper presented at the Waste Management in Antarctica.

Bleasel, J. E. (1989). Waste disposal in the Antarctic. *Report of the SCAR Panel of Experts on Waste Disposal*. Tasmania, Australia: Australian Antarctic Division for SCAR.

Bruni, V., Maugeri, T. L., & Monticelli, L. (1997). Faecal pollution indicators in the Terra Nova Bay (Ross Sea, Antarctica). *Marine Pollution Bulletin, 34*(11), 908-912.

Canadian Council of Ministers of the Environment. (1999). Canadian Soil Quality Guidelines for the Protection of Environmental and Human Health *Canadian Environmental Quality Guidelines*. Winnipeg.

Chapman, P. M., & Riddle, M. J. (2003). Missing and needed: Polar marine ecotoxicology. *Marine Pollution Bulletin, 46*(8), 927-928.

Claridge, G., Campbell, I. B., Powell, H., Amin, Z., & Balks, M. R. (1995). Heavy metal contamination in some soils of the McMurdo Sound region, Antarctica. *Antarctic Science, 7*(1), 9-14.

Clark, D., Lamare, M., & Barker, M. (2009). Response of sea urchin pluteus larvae (Echinodermata: Echinoidea) to reduced seawater pH: a comparison among a tropical, temperate, and a polar species. *Marine Biology, 156*(6), 1125-1137.

Clesceri, L. S., Greenberg, A. E., & Eaton, A. D. (1998). *Standard Methods for the Examination of Water & Wastewater* (20th ed.): American Public Health Association.

COMNAP. (2005). *Working Paper on "Practical Guidelines for Developing and Designing Environmental Monitiroing Programmes in Antarctica"*. Stockholm: XXVIII Antarctic Treaty Consulative Meeting.

Conlan, K. E., Kim, S. L., Lenihan, H. S., & Oliver, J. S. (2004). Benthic changes during 10 years of organic enrichment by McMurdo Station, Antarctica. *Marine Pollution Bulletin, 49*(1-2), 43-60.

Conlan, K. E., Kim, S. L., Thurber, A. R., & Hendrycks, E. (2010). Benthic changes at McMurdo Station, Antarctica following local sewage treatment and regional iceberg-mediated productivity decline. *Marine Pollution Bulletin, 60*(3), 419-432.

Connor, M. A. (2008). Wastewater treatment in Antarctica. *Polar Record, 44*(02), 165-171.

Convey, P., Gibson, J. A. E., Hillenbrand, C. D., Hodgson, D. A., Pugh, P. J. A., Smellie, J. L., & Stevens, M. I. (2008). Antarctic terrestrial life - Challenging the history of the frozen continent? *Biological Reviews, 83*(2), 103-117.

Crockett, A. B. (1997). Water and wastewater quality monitoring, McMurdo Station, Antarctica. *Environmental Monitoring and Assessment, 47*(1), 39-57.

Crockett, A. B., & White, G. J. (2003). Mapping sediment contamination and toxicity in Winter Quarters Bay, McMurdo Station, Antarctica. *Environmental Monitoring and Assessment, 85*(3), 257-275.

Cunningham, L., Raymond, B., Snape, I., & Riddle, M. J. (2005). Benthic diatom communities as indicators of anthropogenic metal contamination at Casey Station, Antarctica. *Journal of Paleolimnology, 33*(4), 499-513.

Curtosi, A., Pelletier, E., Vodopivez, C., St Louis, R., & Mac Cormack, W. P. (2010). Presence and distribution of persistent toxic substances in sediments and marine organisms of Potter Cove, Antarctica. *Archives of Environmental Contamination and Toxicology, 59*(4), 582-592.

Danulat, E., Muniz, P., Garcia-Alonso, J., & Yannicelli, B. (2002). First assessment of the highly contaminated harbour of Montevideo, Uruguay. *Marine Pollution Bulletin, 44*(6), 551-576.

Dayton, P. K., & Robilliard, G. A. (1971). Implications of pollution to the McMurdo Sound benthos. *Antarctic Journal of the United States, 6*(3), 53-56.

De Baar, H. J. W., de Jong, J. T. M., Bakker, D. C. E., Löscher, B. M., Veth, C., Bathmann, U., & Smetacek, V. (1995). Importance of iron for plankton blooms and carbon dioxide drawdown in the Southern Ocean. *Nature, 373*, 412 - 415.

De Moreno, J. E. A., Gerpe, M. S., Moreno, V. J., & Vodopivez, C. (1997). Heavy metals in Antarctic organisms. *Polar Biology, 17*(2), 131-140.

Delille, D., & Delille, E. (2000). Distribution of Enteric Bacteria in Antarctic Seawater Surrounding the Dumont d'Urville Permanent Station (Adelie Land). *Marine Pollution Bulletin, 40*(10), 869-872.

Dingwall, P. R. (1998). Environmental management for Antarctic wilderness. In P. R. Dingwall (Ed.), *Antarctica in the environmental era* (Vol. 1, pp. 1). Wellington: Department of Conservation.

Duquesne, S., Riddle, M., Schulz, R., & Liess, M. (2000). Effects of contaminants in the Antarctic environment - Potential of the gammarid amphipod crustacean Paramorea walkeri as a biological indicator for Antarctic ecosystems based on toxicity and bioacccumulation of copper and cadmium. *Aquatic Toxicology, 49*(1-2), 131-143.

Elliott, C. (2005). Antarctica, Scott Base and its environs. *New Zealand Geographer, 61*(1), 68-76.

Ellis-Evans, J., Laybourn-Parry, J., Bayliss, P., & Perriss, S. (1997). Human impact on an oligotrophic lake in the Larsemann Hills. *Antarctic Communities: Species, Structure and Survival.*, 396-404.

Emara, H. I. (1998). Total organic carbon content in the waters of the Arabian Gulf. *Environment international, 24*(1-2), 97-103.

EPA. (1999). *Wastewater Technology Fact Sheet: Ultraviolet Disinfection*. Washington D.C.

EPA. (2009). Drinking Water Act (SDWA). *National Recommended Water Quality Criteria*.

Evans, C., Hills, J., & Dickson, J. (2000). Heavy metal pollution in Antarctica: a molecular ecotoxicological approach to exposure assessment. *Journal of Fish Biology, 57*, 8-19.

Fitzwater, S. E., Johnson, K. S., Gordon, R. M., Coale, K. H., & Smith W.O, Jr. (2000). Trace metal concentrations in the Ross Sea and their relationship with nutrients and phytoplankton growth. *Deep-Sea Research Part II: Topical Studies in Oceanography, 47*(15-16), 3159-3179.

Fox, A. J., & Cooper, A. P. R. (1994). Measured properties of the Antarctic ice sheet derived from the SCAR Antarctic digital database. *Polar Record, 30*(174), 201-206.

Frache, R., Abelmoschi, M. L., Grotti, M., Ianni, C., Magi, E., Soggia, F., Capodaglio, G., Turetta, C., & Barbante, C. (2001). Effects of ice melting on Cu, Cd and Pb profiles in Ross Sea waters (Antarctica). *International Journal of Environmental Analytical Chemistry, 79*(4), 301-313.

Gardner, H., Kerry, K., Riddle, M., Brouwer, S., & Gleeson, L. (1997). Poultry virus infection in Antarctic penguins. *Nature, 387*(6630), 245.

GERG. (2000). *Antarctic Environmental Monitoring Handbook: Standard techniques for monitoring in Antarctica*. Council of Managers of National Antarctic Programs (COMNAP), Scientific Committee on Antarctic Research (SCAR).

Ghosh, T. K., Muley, R. D., Ghode, R., & Ramteke, D. S. (1997). Scientific Report of the Thirteenth Indian expedition to Antarctica. *Department of Ocean Development, Technical Publication No. 11*, 301-311.

Gogate, P. R., & Pandit, A. B. (2004). A review of imperative technologies for wastewater treatment I: oxidation technologies at ambient conditions. *Advances in Environmental Research, 8*(3–4), 501-551.

Goldsworthy, P. M., Canning, E. A., & Riddle, M. J. (2003). Soil and water contamination in the Larsemann Hills, East Antarctica. *Polar Record, 39*(211), 319-337.

Green, G., & Nichols, P. D. (1995). Hydrocarbons and sterols in marine sediments and soils at Davis Station, Antarctica: a survey for human-derived contaminants. *Antarctic Science, 7*(02), 137-144.

Grimaldi, W., Jabour, J., & Woehler, E. J. (2011). Considerations for minimising the spread of infectious disease in Antarctic seabirds and seals. *Polar Record, 47*(1), 56-66.

Grotti, M., Soggia, F., Abelmoschi, M. L., Rivaro, P., Magi, E., & Frache, R. (2001). Temporal distribution of trace metals in Antarctic coastal waters. *Marine Chemistry, 76*(3), 189-209.

Grotti, M., Soggia, F., Ianni, C., & Frache, R. (2005). Trace metals distributions in coastal sea ice of Terra Nova Bay, Ross Sea, Antarctica. *Antarctic Science, 17*(2), 289-300.

Gröndahl, F., Sidenmark, J., & Thomsen, A. (2009). Survey of waste water disposal practices at Antarctic research stations. *Polar Research, 28*(2), 298-306.

Hale, R. C., Kim, S. L., Harvey, E., La Guardia, M. J., Mainor, T. M., Bush, E. O., & Jacobs, E. M. (2008). Antarctic research bases: local sources of polybrominated diphenyl ether (PBDE) flame retardants. *Environmental science & technology, 42*(5), 1452-1457.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C., Fox, H. E., Fujita, R., Heinemann, D., Lenihan, H. S., Madin, E. M. P., Perry, M. T., Selig, E. R., Spalding, M., Steneck, R., & Watson, R. (2008). A global map of human impact on marine ecosystems. *Science, 319*(5865), 948-952.

Harris, J. E., & Fabris, G. J. (1979). Concentrations of suspended matter and particulate cadmium, copper, lead and zinc in the Indian sector of the Antarctic Ocean. *Marine Chemistry, 8*(2), 163-179.

Harrowfield, D. L. (1997). *Scott Base, Antarctica: a history of New Zealand's southern-most station, 1957-1997 ; commemorating the 40th anniversary of Scott Base*. Christchurch, N.Z.: New Zealand Antarctic Society.

Heaton, K., & Paterson, C. (2003). Design of a wastewater treatment plant for Davis base Antarctica. *Water(Australia), 30*(4), 78-82.

Hendry, K. R., Rickaby, R. E. M., de Hoog, J., Weston, K., & Rehkämper, M. (2008). Cadmium and phosphate in coastal Antarctic seawater: Implications for Southern Ocean nutrient cycling. *Marine Chemistry, 112*(3-4), 149-157.

Hernandez, J., Prado, V., Torres, D., Waldenström, J., Haemig, P. D., & Olsen, B. (2007). Enteropathogenic Escherichia coli (EPEC) in Antarctic fur seals Arctocephalus gazella. *Polar Biology, 30*(10), 1227-1229.

Howington, J. P., McFeters, G. A., Barry, J. P., & Smith, J. J. (1992). Distribution of the McMurdo Station sewage plume. *Marine Pollution Bulletin, 25*(9-12), 324-327.

Hughes, K. A. (2003). Influence of seasonal environmental variables on the distribution of presumptive fecal coliforms around an Antarctic research station. *Applied and Environmental Microbiology, 69*(8), 4884.

Hughes, K. A. (2004). Reducing sewage pollution in the Antarctic marine environment using a sewage treatment plant. *Marine Pollution Bulletin, 49*(9-10), 850-853.

Hughes, K. A. (2005). Effect of Antarctic solar radiation on sewage bacteria viability. *Water Research, 39*(11), 2237-2244.

Hughes, K. A. (2010). How committed are we to monitoring human impacts in Antarctica? *Environmental Research Letters, 5*, 041001.

Hughes, K. A., & Blenkharn, N. (2003). A simple method to reduce discharge of sewage microorganisms from an Antarctic research station. *Marine Pollution Bulletin, 46*(3), 353-357.

Hughes, K. A., & Thompson, A. (2004). Distribution of sewage pollution around a maritime Antarctic research station indicated by faecal coliforms, Clostridium perfringens and faecal sterol markers. *Environmental Pollution 127*(3), 315-321.

Illuminati, S., Truzzi, C., Annibaldi, A., Migliarini, B., Carnevali, O., & Scarponi, G. (2010). Cadmium bioaccumulation and metallothionein induction in the liver of the antarctic teleost trematomus bernacchii during an on-site short-term exposure to the metal via seawater. *Toxicological and Environmental Chemistry, 92*(3), 617-640.

Kennicutt II, M. C., Klein, A., Montagna, P., Sweet, S., Wade, T., Palmer, T., Sericano, J., & Denoux, G. (2010). Temporal and spatial patterns of anthropogenic disturbance at McMurdo Station, Antarctica. *Environmental Research Letters, 5*, 034010.

Kennicutt II, M. C., McDonald, S., Sericano, J., Boothe, P., Oliver, J., Safe, S., Presley, B., Liu, H., Wolfe, D., & Wade, T. (1995). Human contamination of the marine environment-Arthur Harbor and McMurdo Sound, Antarctica. *Environmental Science & Technology, 29*(5), 1279-1287.

King, C., & Riddle, M. (2001). Effects of metal contaminants on the development of the common Antarctic sea urchin Sterechinus neumayeri and comparisons of sensitivity with tropical and temperate echinoids. *Marine Ecology Progress Series, 215*, 143-154.

Kramer, D. L. (1987). Dissolved oxygen and fish behavior. *Environmental Biology of Fishes, 18*(2), 81-92.

Lannuzel, D., Bowie, A. R., van der Merwe, P. C., Townsend, A. T., & Schoemann, V. (2011). Distribution of dissolved and particulate metals in Antarctic sea ice. *Marine Chemistry, 124*(1-4), 134-146. doi: 10.1016/j.marchem.2011.01.004

Leitch, J. (2011). [Scott Base Wastewater Treatment Plant].

Lenihan, H. S., & Oliver, J. S. (1995). Anthropogenic and natural disturbances to marine benthic communities in Antarctica. *Ecological Applications, 5*(2), 311-326.

Lenihan, H. S., Oliver, J. S., Oakden, J. M., & Stephenson, M. D. (1990). Intense and localized benthic marine pollution around McMurdo Station, Antarctica. *Marine Pollution Bulletin, 21*(9), 422-430.

Lisle, J. T., Smith, J. J., Edwards, D. D., & McFeters, G. A. (2004). Occurrence of microbial indicators and Clostridium perfringens in wastewater, water column samples, sediments, drinking water, and Weddell seal feces collected at McMurdo Station, Antarctica. *Applied and Environmental Microbiology, 70*(12), 7269-7276.

Luker, M. A. (2009). *Testing metal mobility in artificially contaminated soils at Scott Base, Antarctica.* Unpublished Thesis for Master of Science, Univiersity of Auckland, Auckland.

Madrid Protocol. (1991). *Protocol on Environmental Protection to the Antarctic Treaty*.

Malandrino, M., Abollino, O., Buoso, S., Casalino, C. E., Gasparon, M., Giacomino, A., La Gioia, C., & Mentasti, E. (2009). Geochemical characterisation of Antarctic soils and lacustrine sediments from Terra Nova Bay. *Microchemical journal, 92*(1), 21-31.

Martin, J. H., Gordon, R. M., & Fitzwater, S. E. (1990). Iron in Antarctic waters. *Nature, 345*(156-158).

McFeters, G. A., Barry, J. P., & Howington, J. P. (1993). Distribution of enteric bacteria in Antarctic seawater surrounding a sewage outfall. *Water Research, 27*(4), 645-650.

Metcalf & Eddy Inc. (2008). United States Antarctic Program; Master Permit Application Retrieved 10 August 2011, 2011, from http://www.nsf.gov/about/contracting/rfqs/support_ant/docs/permit_application_text_tagged.pdf

Miller, H. C., Mills, G. N., Bembo, D. G., Macdonald, J. A., & Evans, C. W. (1999). Induction of cytochrone P4501A (CYP1A) in *Trematomus bernacchii* as an indicator of environmental pollution in Antarctica: assessment by quantitative RT-PCR. *Aquatic Toxicology, 44*, 183-193.

Ministry of Health. (2008). Drinking-water standards for New Zealnd 2005 (Revised 2008). *Wellington: Ministry of Health*.

Montone, R. C., Martins, C. C., Bicego, M. C., Taniguchi, S., Albuquerque Moreira da Silva, D., Campos, L. S., & Weber, R. R. (2010). Distribution of sewage input in marine sediments around a maritime Antarctic research station indicated by molecular geochemical indicators. *Science of the Total Environment, 408*(20), 4665-4671.

Morris, C. E., George, J., Tate, P. M., & Cathers, B. (2000). *Impacts of wastewater discharges to the Antarctic marine environment* Paper presented at the Proceedings of the Sixth International Symposium on Cold Regions Development, Hobart.

National Institute of Standards and Technology. (2004). Certificate of Analysis: Standard Reference Material 2702 (Inorganics in Marine Sediment). *Department of Commerce, United States of America*.

Negri, A., Burns, K., Boyle, S., Brinkman, D., & Webster, N. (2006). Contamination in sediments, bivalves and sponges of McMurdo Sound, Antarctica. *Environmental Pollution, 143*(3), 456-467.

Nolting, R., & De Baar, H. (1994). Behaviour of nickel, copper, zinc and cadmium in the upper 300 m of a transect in the Southern Ocean. *Marine Chemistry, 45*(3), 225-242.

Owili, M. A. (2003). Assesment of Impact of Sewage Effluents on Coastal Water Quality in Hafnarfjordur, Iceland. Reykjavik, Iceland: Icelandic Fisheries Laboratories.

Papoff, P., Bocci, F., & Onor, M. (1996). Distribution of Macro-and Macro-Components in the Water Column of the Antarctic Ross Sea and in Surface Antarctic Snow. *International Journal of Environmental Analytical Chemistry, 63*(1), 1-13.

Pearce, D. A., & Wilson, W. H. (2003). Viruses in Antarctic ecosystems. *Antarctic Science, 15*(03), 319-331.

Rack, W. (Cartographer). (2012). Aerial photograph of Scott Base, Antarctica.

Railsback, S. F. (1992). Mixing characteristics of submerged and surface wastewater outfalls at McMurdo Station. *Polar Record, 28*(165), 149-154.

Raytheon Polar Services. (2007). Operations and Maintenance Manual: Wastewater Treatment Plant Building 199, McMurdo Research Station, Antarctica. *National Science Foundation Office of Polar Programs*.

Redvers, G. (2000). *Dispersion and Fate of Sewage and Wastewater Components From Scott Base, Antarctica.* Unpublished Thesis for Master of Science, University of Auckland, Auckland.

Reinthaler, F. F., Posch, J., Feierl, G., Wüst, G., Haas, D., Ruckenbauer, G., Mascher, F., & Marth, E. (2003). Antibiotic resistance of E. coli in sewage and sludge. *Water Research, 37*(8), 1685-1690.

Ribeiro, A. P., Figueira, R. C. L., Martins, C. C., Silva, C. R. A., França, E. J., Bícego, M. C., Mahiques, M. M., & Montone, R. C. (2011). Arsenic and trace metal contents in sediment profiles from the Admiralty Bay, King George Island, Antarctica. *Marine Pollution Bulletin, 62*(1), 192-196.

Sanchez - Hernandez, J. C. (2000). Trace Element Contamination in Antarctic Ecosystems. *Reviews of Environmental Contamination and Toxicology, 166*, 83-127.

Sawyer, C. N., McCarty, P. L., & Parkin, G. F. (1994). *Chemistry for Environmental Engineering*. New York: McGraw-Hill, Inc.

Scarponi, G., Capodaglio, G., Toscano, G., Barbante, C., & Cescon, P. (1995). Speciation of lead and cadmium in Antarctic seawater: comparison with areas subject to different anthropic influence. *Microchemical journal, 51*(1-2), 214-230.

Secretariant of the Antarctic Treaty. (1959). *The Antarctic Treaty*. Paper presented at the Conference on Antarctica, Washington, D.C.

Sheppard, D. S., Claridge, G., & Campbell, I. (2000). Metal contamination of soils at Scott Base, Antarctica. *Applied Geochemistry, 15*(4), 513-530.

Sheppard, D. S., Deely, J. M., & Edgerley, W. H. L. (1997). Heavy metal content of meltwaters from the Ross Dependency, Antarctica. *New Zealand Journal of Marine and Freshwater Research, 31*(3), 313-325.

Smith, J. J., Howington, J. P., & McFeters, G. A. (1994). Survival, physiological response and recovery of enteric bacteria exposed to a polar marine environment. *Applied and Environmental Microbiology, 60*(8), 2977-2984.

Smith, J. J., & Riddle, M. J. (2009). Sewage Disposal and Wildlife Health in Antarctica. In K. R. Kerry & M. Riddle (Eds.), *Health of Antarctic Wildlife* (pp. 271-315): Springer Berlin Heidelberg.

Snape, I., Acomb, L., Barnes, D. I., Bainbridge, S., Eno, R., Filler, M., Plato, N., Raymond, T., Rayner, J. L., Riddle, M. J., Rike, A. G., Rutter, A., Schafer, A. L., Siciliano, S. D., & Walworth, J. L. (2008). Contamination, regulation and remediation: an introduction to bioremediation of petroleum hydrocarbons in cold regions. In D. M. Filler, I. Snape & D. L. Barnes (Eds.), *Bioremediation of petroleum hydrocarbons in cold regions* (pp. 1-37). Cambridge Cambridge University Press.

Stark, J. S., Riddle, M. J., & Simpson, R. D. (2003a). Human impacts in soft-sediment assemblages at Casey Station, East Antarctica: Spatial variation, taxonomic resolution and data transformation. *Austral Ecology, 28*(3), 287-304.

Stark, J. S., Snape, I., & Riddle, M. J. (2003b). The effects of petroleum hydrocarbon and heavy metal contamination of marine sediments on recruitment of Antarctic soft-sediment assemblages: a field experimental investigation. *Journal of Experimental Marine Biology and Ecology, 283*(1-2), 21-50.

Statham, J. A., & McMeekin, T. A. (1994). Survival of faecal bacteria in Antarctic coastal waters. *Antarctic Science, 6*(03), 333-338.

Stern, B. R. (2010). Essentiality and toxicity in copper health risk assessment: overview, update and regulatory considerations. *Journal of Toxicology and Environmental Health, Part A, 73*(2-3), 114-127.

Suttie, E., & Wolff, E. (1993). The local deposition of heavy metal emissions from point sources in Antarctica. *Atmospheric Environment. Part A. General Topics, 27*(12), 1833-1841.

Tin, T., Fleming, Z. L., Hughes, K. A., Ainley, D. G., Convey, P., Moreno, C. A., Pfeiffer, S., Scott, J., & Snape, I. (2008). Review: Impacts of local human activities on the Antarctic environment. *Antarctic Science, 21*(1), 3-33.

Townsend, A. T., & Snape, I. (2008). Multiple Pb sources in marine sediments near the Australian Antarctic Station, Casey. *Science of the Total Environment, 389*(2), 466-474.

US EPA. (1994). *Method 1312: Synthetic Precipitation Leaching Procedure*.

Van Ngan, P., Gomes, V., Passos, M., Ussami, K., Campos, D., Rocha, A., & Pereira, B. (2007). Biomonitoring of the genotoxic potential (micronucleus and erythrocyte nuclear abnormalities assay) of the Admiralty Bay water surrounding the Brazilian Antarctic Research Station "Comandante Ferraz," King George Island. *Polar Biology, 30*(2), 209-217.

Visco, G., Campanella, L., & Nobili, V. (2005). Organic carbons and TOC in waters: an overview of the international norm for its measurements. *Microchemical journal, 79*(1), 185-191.

Walton, D. W. H. (1998). Environmental monitoring in Antarctica - measuring the damage. In P. R. Dingwall (Ed.), *Antarctica in the Environmental Era* (Vol. 1, pp. 9). Wellington: Department of Conservation.

Waterhouse, E., J. (Ed.). (2001). *Ross Sea Region 2001: A State of the Environment Report for the Ross Sea Region of Antarctica* New Zealand Antarctic Institute, Christchurch, New Zealand.

Watier, C. (2008). *Environmental Monitoring at Swedish Research Stations in Antarctica.* Master of Science Royal Institute of Technology, Sweden.

Webster-Brown, J. G. (2012). [Calibration: FAU units against NTU units].

Westerlund, S., & Ohman, P. (1991). Cadmium, copper, cobalt, nickel, lead, and zinc in the water column of the Weddell Sea, Antarctica. *Geochimica et Cosmochimica Acta, 55*(8), 2127-2146.

Wolff, E. (1990). Signals of atmospheric pollution in polar snow and ice. *Antarctic Science, 2*(03), 189-205.

Wood, I. R., Bell, R. G., & Wilkinson, D. L. (1993) Ocean disposal of wastewater. *Vol. 8. Advanced Series on Ocean Engineering*: World Scientific.

8 Appendix

Analytical Seawater Data

Scott Base Field Sites

Site	Depth (m)	Nitrate - Nitrogen (mg/L)	Phosphorus (mg/L)	Suspended Solids (mg/L)	pH	DO (mg/L)	Conductivity (mS/cm)
1	1.0	<0.8	0.38	12.2	7.89	11.8	52.2
	6.0	<0.8	0.42	11.9	7.91	12.01	52.3
	12.5	<0.8	0.38	10.6	7.86	12.08	52.8
2	1.0	<0.8	0.69	13.9	7.83	11.96	48.9
	13.0	<0.8	0.33	16.4	7.77	12.72	50.4
	25.0	<0.8	0.28	12.4	7.82	12.17	51.9
3	1.0	<0.8	0.34	12.1	7.92	12.44	51.5
	18.0	<0.8	0.49	12	7.92	12.3	48.4
	35.0	<0.8	0.38	9	7.9	12.88	52.2
4	1.0	<0.8	0.34	12.2	7.83	12.7	53.0
	25.0	<0.8	0.46	13	7.84	12.38	52.0
	55.0	<0.8	0.47	12.4	7.83	12.54	52.9
5	1.0	<0.8	0.44	11.5	7.81	11.52	53.3
	17.0	<0.8	0.37	11.3	7.82	11.61	53.3
	33.0	<0.8	0.33	15.3	7.79	11.26	53.6
6	1.0	<0.8	0.46	15.8	7.83	11.69	51.8
	7.0	<0.8	0.41	14.4	7.87	11.76	52.7
	14.0	<0.8	0.41	12.2	7.82	11.92	52.8
8	1.0	<0.8	0.36	12.8	7.88	11.68	52.5
	5.0	<0.8	0.49	13.7	7.87	11.63	53.1
	10.0	<0.8	0.28	16.5	7.84	11.53	52.9

Scott Base Field Sites - continued

Site	Depth (m)	Nitrate - Nitrogen (mg/L)	Phosphorus (mg/L)	Suspended Solids (mg/L)	pH	DO (mg/L)	Conductivity (mS/cm)
9	1.0	<0.8	0.36	11	7.85	12.6	53.0
	8.0	1	0.64	12	7.9	12.8	51.0
	17.0	1.1	0.57	12.2	7.42	12.39	55.2
10	1.0	<0.8	0.46	14.9	7.8	12.46	52.3
	10.0	<0.8	0.41	12.7	7.85	12.72	49.6
	20.0	<0.8	0.54	14.7	7.84	12.51	51.4
11	1.0	<0.8	0.41	13.5	7.8	11.12	53.3
	17.0	<0.8	0.38	12.5	7.8	11.45	50.8
	25.0	<0.8	0.53	10.7	7.85	11.41	53
12	1.0	0.8	0.47	10.5	7.72	10.05	54.6
13	1.0	<0.8	0.54	12.9	7.81	11.68	52.9
	5.0	<0.8	0.49	10.9	7.82	11.6	53.4
	10.0	<0.8	0.58	10.9	7.8	11.2	54.3
14	1.0	1.1	0.38	14.7	7.85	11.93	52.4
	12.0	<0.8	0.43	12.1	7.82	12.29	53.1
	25.0	<0.8	0.45	13	7.86	12.24	52.8
15	1.0	<0.8	0.44	16.6	7.84	11.58	51.6
	6.0	<0.8	0.8	14.8	7.82	12.34	53.3
18	1.0	<0.8	0.38	13.1	7.88	12.95	51.6
	25.0	<0.8	0.45	17.8	7.88	12.57	51.7
	50.0	<0.8	0.71	16.3	7.87	13.02	52.6

Scott Base Field Sites - Total Trace Metals

	Site	Depth (m)	Li (µg/L)	Mn (µg/L)	Fe (µg/L)	Ni (µg/L)	Cu (µg/L)	Zn (µg/L)	As (µg/L)	Cd (µg/L)	Pb (µg/L)
Total	4	1	198.16	<0.34	16.9	1.11	<1.2	<12	2.81	0.29	<6
		25	194.35	<0.34	22.3	1.28	2.67	<12	2.94	0.28	<6
		55	197.64	0.42	29.6	0.97	<1.2	<12	2.90	0.31	<6
	5	40	<0.59	0.36	28.3	0.55	<1.2	<12	2.80	0.09	<6
	8	10	<0.59	0.34	26.2	0.62	<1.2	<12	3.66	0.05	<6
	9	1	195.55	<0.34	<16.8	0.41	<1.2	<12	2.73	0.10	<6
		9	191.56	<0.34	17.4	1.12	<1.2	<12	2.98	0.35	<6
		17	193.87	<0.34	27.9	1.29	<1.2	<12	2.71	0.30	<6
	10	20	<0.59	0.38	20.9	0.56	<1.2	<12	3.31	0.06	<6
	12	1	204.32	0.42	<16.8	0.67	<1.2	<12	2.97	0.11	<6
	13	1	198.73	1.32	18.1	1.13	2.10	34.92	6.77	0.36	<6
		5	190.93	<0.34	20.5	1.28	<1.2	<12	2.91	0.33	<6
		10	199.19	<0.34	26.4	0.92	<1.2	<12	3.00	0.36	<6
	14	1	197.92	<0.34	18.3	0.90	<1.2	<12	2.83	0.36	<6
		12	193.85	0.55	64.2	0.78	2.54	<12	2.91	0.34	<6
		25	195.84	<0.34	<16.8	0.72	<1.2	<12	2.76	0.09	<6
	15	1	193.80	<0.34	<16.8	1.12	<1.2	<12	2.67	0.31	<6
		6	185.68	2.23	35.0	1.73	26.49	27.18	2.67	0.37	<6
	18	1	198.61	<0.34	18.5	1.03	<1.2	<12	2.73	0.27	<6
		25	203.02	<0.34	20.2	1.00	<1.2	<12	2.94	0.30	<6
		50	199.32	<0.34	<16.8	0.43	<1.2	<12	2.95	0.22	<6

Scott Base Field Sites - Dissolved Trace Metals

	Site	Depth (m)	Li (µg/L)	Mn (µg/L)	Fe (µg/L)	Ni (µg/L)	Cu (µg/L)	Zn (µg/L)	As (µg/L)	Cd (µg/L)	Pb (µg/L)
Dissolved	4	1	188.20	<0.34	<16.8	0.57	<1.2	<12	2.60	0.27	<6
		25	196.06	<0.34	<16.8	0.65	<1.2	<12	2.55	0.27	<6
		55	186.49	<0.34	<16.8	0.65	<1.2	<12	2.61	0.28	<6
	5	40	<0.59	<0.34	<16.8	<0.43	<1.2	<12	2.80	0.10	<6
	8	10	<0.59	<0.34	<16.8	0.71	<1.2	<12	3.13	0.12	<6
	9	1	197.71	<0.34	<16.8	0.77	<1.2	<12	2.99	0.16	<6
		9	206.60	<0.34	<16.8	0.62	<1.2	<12	2.63	0.29	<6
		17	196.45	<0.34	<16.8	1.05	<1.2	<12	2.59	0.27	<6
	10	20	<0.59	<0.34	<16.8	<0.43	<1.2	<12	3.03	0.11	<6
	12	1	192.55	<0.34	<16.8	0.48	<1.2	<12	2.57	0.22	<6
	13	1	192.12	1.21	<16.8	1.07	3.14	31.25	6.14	0.40	<6
		5	198.39	<0.34	17.6	0.74	<1.2	<12	2.87	0.26	<6
		10	194.19	<0.34	<16.8	0.62	<1.2	<12	2.79	0.30	<6
	14	1	201.98	<0.34	<16.8	0.58	<1.2	<12	2.94	0.32	<6
		12	192.22	<0.34	<16.8	0.60	<1.2	<12	2.51	0.30	<6
		25	193.74	<0.34	<16.8	0.71	-1.06	<12	2.65	0.10	<6
	15	1	189.48	<0.34	<16.8	0.61	<1.2	<12	2.96	0.28	<6
		6	194.69	1.72	<16.8	1.01	22.23	28.45	2.55	0.32	<6
	18	1	202.08	<0.34	<16.8	0.60	<1.2	<12	2.78	0.34	<6
		25	199.49	<0.34	<16.8	0.82	<1.2	<12	2.92	0.29	<6
		50	185.33	<0.34	<16.8	0.51	<1.2	<12	2.94	<0.05	<6

Scott Base WWTP and Drinking Water - Trace Metals

	Site	Sample Run	Li (µg/L)	Mn (µg/L)	Fe (µg/L)	Ni (µg/L)	Cu (µg/L)	Zn (µg/L)	As (µg/L)	Cd (µg/L)	Pb (µg/L)
Dissolved	WWTP Pre-Treatment	1	4.83	39.23	233.4	9.09	5.71	53.10	1.35	0.39	<6
		2	4.68	36.05	196.9	7.84	4.53	45.36	1.17	0.14	<6
	WWTP Post-Treatment	1	4.61	56.04	258.1	10.53	6.00	59.68	1.51	0.16	<6
		2	4.74	30.33	190.6	8.77	7.23	50.18	1.25	0.16	<6
		3	4.69	34.19	150.4	9.77	10.43	66.40	1.31	0.16	<6
	RO intake	1	200.18	0.98	<16.8	72.44	25.82	429.96	2.50	2.97	<6
		2	193.58	<0.34	<16.8	12.55	5.85	145.43	2.68	0.84	<6
	RO2	1	0.55	<0.34	<16.8	0.92	94.17	77.18	<0.56	0.24	<6
		2	0.49	<0.34	<16.8	0.70	91.21	60.92	<0.56	0.22	<6
	RO1	1	2.79	1.82	<16.8	43.81	37.87	289.70	<0.56	0.41	9.68
		2	2.79	1.62	<16.8	27.43	25.00	286.39	<0.56	0.45	18.97
Total	WWTP Pre-Treatment	1	4.34	43.84	255.0	10.25	5.95	62.88	1.49	0.15	<6
		2	4.93	37.93	203.2	8.92	6.04	50.49	1.26	0.18	<6
	WWTP Post-Treatment	1	5.09	59.23	257.2	10.98	8.52	60.75	1.37	0.21	<6
		2	5.53	33.19	186.0	9.27	6.82	62.52	1.33	0.19	<6
		3	4.23	34.55	160.7	10.02	11.62	68.68	1.27	0.20	<6
	RO intake	1	205.98	1.12	<16.8	80.15	31.21	454.14	2.69	3.14	<6
		2	202.15	<0.34	<16.8	13.96	6.53	155.04	2.95	0.84	<6
	RO2	1	<0.59	<0.34	23.3	1.62	96.50	74.84	<0.56	0.24	<6
		2	<0.59	<0.34	17.4	1.30	101.20	75.82	<0.56	0.25	<6
	RO1	1	2.29	1.85	<16.8	45.34	44.61	311.28	<0.56	0.51	<6
		2	2.75	1.69	21.4	29.17	26.49	313.95	<0.56	0.56	<6

Winter Quarters Bay Field Sites

Site	Depth (m)	Nitrate - Nitrogen (mg/L)	Phosphorus (mg/L)	Suspended Solids (mg/L)	pH	DO (mg/L)	Conductivity (mS/cm)
1	1	<0.8	0.46	14.6	7.88	12.69	49.8
	2	0.9	0.84	12	7.85	13.12	50.7
2	1	<0.8	0.95	11.6	7.86	12.54	51.2
	9	<0.8	0.35	12	7.87	12.96	51.9
	18	<0.8	0.58	18.4	7.94	13.15	51.3
3	1	<0.8	0.58	11.4	7.83	12.88	52.4
	15	<0.8	0.44	17.2	7.86	12.84	51.8
	30	<0.8	0.34	12.2	7.84	12.56	51.5

Winter Quarters Bay - Trace Metals

	Site	Depth (m)	Li (µg/L)	Mn (µg/L)	Fe (µg/L)	Ni (µg/L)	Cu (µg/L)	Zn (µg/L)	As (µg/L)	Cd (µg/L)	Pb (µg/L)
Dissolved	1	1	211.59	<0.34	<16.8	0.71	<1.2	<12	2.90	0.29	<6
		2	191.13	<0.34	<16.8	0.48	3.30	<12	2.74	0.28	<6
	2	1	198.99	<0.34	17.4	0.71	<1.2	<12	2.64	0.33	<6
		9	200.60	<0.34	16.2	0.84	<1.2	<12	2.74	0.28	<6
		18	200.06	<0.34	<16.8	0.70	<1.2	<12	2.68	0.29	<6
	3	1	193.95	<0.34	<16.8	0.67	<1.2	<12	2.84	0.30	<6
		15	198.41	<0.34	<16.8	0.67	<1.2	13.16	2.81	0.28	<6
		30	198.82	<0.34	<16.8	0.65	<1.2	<12	3.38	0.29	<6
Total	1	1	193.94	0.34	20.6	0.83	<1.2	<12	2.75	0.31	<6
		2	<0.59	<0.34	<16.8	<0.43	<1.2	<12	<0.56	0.15	<6
	2	1	199.77	<0.34	20.9	0.77	<1.2	<12	2.85	0.40	<6
		9	199.83	0.47	38.2	651.78	6.61	1625.11	3.50	0.39	10.94
		18	182.97	0.41	21.6	1.45	<1.2	<12	2.61	0.29	<6
	3	1	196.60	<0.34	22.7	0.74	<1.2	<12	2.89	0.33	<6
		15	196.35	<0.34	20.1	0.82	<1.2	<12	2.89	0.30	<6
		30	202.43	<0.34	21.6	0.99	1.20	<12	2.76	0.31	<6

Control Site

Site	Depth (m)	Nitrate - Nitrogen (mg/L)	Phosphorus (mg/L)	Suspended Solids (mg/L)	pH	DO (mg/L)	Conductivity (mS/cm)
Control	5	0.4	0.43	13.8	7.77	11.71	53
	15	0.4	0.62	11	7.82	11.63	52
	30	0.4	0.52	15.4	7.83	11.88	52.8

Control Site - Trace Metals

	Depth (m)	Li (µg/L)	Mn (µg/L)	Fe (µg/L)	Ni (µg/L)	Cu (µg/L)	Zn (µg/L)	As (µg/L)	Cd (µg/L)	Pb (µg/L)
Dissolved	5	201.15	<0.34	<16.8	0.63	<1.2	<12	2.55	0.28	<6
	15	202.27	<0.34	<16.8	0.86	<1.2	<12	3.01	0.22	<6
	30	191.50	<0.34	34.1	0.53	<1.2	<12	2.51	0.32	<6
Total	5	198.72	<0.34	26.0	1.21	<1.2	<12	2.89	0.30	<6
	15	<0.59	<0.34	<16.8	<0.43	<1.2	<12	<0.56	0.13	<6
	30	191.77	0.60	50.6	1.55	<1.2	19.23	2.86	0.37	<6

Transport and Field Trace Metal Blanks

		Sample	Mn (µg/L)	Fe (µg/L)	Ni (µg/L)	Cu (µg/L)	Zn (µg/L)	As (µg/L)	Cd (µg/L)	Pb (µg/L)	Ca (µg/L)
Transport Blanks	Dissolved	1	0.56	<16.8	<0.43	<1.2	<12	<0.56	<0.05	<6	46.21
		2	1.13	<16.8	<0.43	<1.2	<12	<0.56	<0.05	<6	126.29
	Total	1	0.90	23.9	<0.43	<1.2	<12	<0.56	<0.05	<6	226.63
		2	<0.34	<16.8	<0.43	2.92	<12	<0.56	<0.05	<6	194.11
Field Blanks	Dissolved	A	<0.34	<16.8	<0.43	<1.2	<12	<0.56	0.40	9.29	17.0083
		B	0.46	<16.8	<0.43	<1.2	<12	<0.56	<0.05	<6	33.12798
		C	<0.34	<16.8	<0.43	<1.2	<12	<0.56	0.08	<6	116.83
	Total	A	<0.34	<16.8	<0.43	<1.2	<12	<0.56	0.13	<6	43.11097
		B	<0.34	<16.8	<0.43	<1.2	<12	<0.56	<0.05	<6	45.45542
		C	0.48	<16.8	<0.43	<1.2	<12	<0.56	<0.05	<6	223.52

Analytical Soil Data

Scott Base Field Sites - Trace Elements in Terrestrial Soil

Sample	Soil Weight	Trace Elements (ug/L)								
	(g)	Cr	Mn	Fe	Ni	Cu	Zn	As	Cd	Pb
S1	1.0452	776.60	23667.02	1136220.83	3337.35	1849.22	1626.26	96.58	3.88	195.72
S2	1.0076	1015.95	26981.50	1243845.83	3106.77	1438.92	2531.98	104.59	7.75	465.11
S3	1.022	834.67	28227.55	1163119.28	2969.10	1383.85	2750.87	115.84	7.14	514.12
S4	1.0326	837.89	34966.02	1274789.80	2927.26	1858.25	4771.26	186.93	15.05	2048.99
S5	1.0013	943.62	29018.33	1221360.06	3227.20	1464.69	2897.00	130.91	5.42	768.67
S6	1.029	1410.19	38879.39	1636334.23	4082.90	2787.35	12116.18	166.39	33.89	4258.35
HP1	1.0376	349.40	17782.80	636342.27	1342.89	840.97	1368.67	65.29	4.63	190.79
HP2	1.0266	340.46	16572.22	658166.18	1702.28	811.12	1480.05	70.35	3.50	160.04
HP3	1.034	585.90	23735.30	934462.76	2599.54	1004.35	3101.12	87.98	4.91	304.68
HP4	1.0134	453.88	21228.34	833437.65	2057.02	984.12	1622.12	81.61	4.96	441.70
HL1	1.0011	745.62	29195.46	1090169.35	3163.81	1926.17	3056.07	165.70	11.35	415.03
HL2	1.0052	898.23	31963.73	1284802.51	3981.80	1328.66	3128.46	124.25	7.95	455.60
HL3	1.009	949.29	25515.60	915449.83	2173.89	3410.79	4764.90	662.18	10.67	427.13
HL4	1.0001	674.21	26020.81	983695.19	2503.15	1298.36	3392.86	123.17	17.76	1590.84
HL5	1.0119	299.57	15386.32	522713.77	1130.72	1298.07	3170.37	65.21	4.76	5182.47
HL6	1.0137	468.39	11784.44	378795.73	734.76	876.84	7957.55	320.37	6.03	395.67
HL7	1.0371	451.40	22895.27	700279.42	1463.95	870.70	1760.31	81.75	6.46	457.21
DT1	1.0103	721.01	25600.47	1109851.11	2968.79	1293.86	1962.88	89.54	4.05	200.48
DT2	1.0111	720.93	25915.98	1129404.61	2945.93	1306.13	1822.42	91.85	4.00	196.20
DT3	1.0233	398.87	17736.62	676473.25	1633.98	923.20	1296.55	67.89	3.37	153.66
DT4	1.0004	432.23	16847.92	686993.52	1863.23	843.02	1346.67	71.41	3.80	257.25
DT5	1.0047	461.88	15899.47	636318.37	1443.90	927.65	1707.47	84.49	6.94	881.44
MG1	1.0012	249.00	11280.29	432399.64	1291.19	559.83	873.95	56.01	3.11	169.52
MG2	1.0086	602.47	20687.18	898699.78	2661.72	899.73	1793.20	75.31	4.77	221.54
MG3	1.0152	442.15	18620.73	719295.14	1935.41	773.09	1614.43	71.01	5.79	307.32

Scott Base Field Sites - Water Leachable Trace Elements

Sample	Soil Weight	Trace Elements (ug/L)								
	(g)	Cr	Mn	Fe	Ni	Cu	Zn	As	Cd	Pb
S1	1.05	<1	1.71	49.9	<0.43	1.46	<12	<0.56	<0.05	<6
S2	1.01	<1	1.61	56.9	<0.43	4.19	<12	0.94	<0.05	<6
S3	1.02	<1	1.95	61.67	0.58	7.86	<12	2.27	<0.05	<6
S4	1.03	<1	21.05	1017.7	1.39	8.67	<12	7.34	<0.05	<6
S5	1.00	1.1	45.90	1891.5	2.43	7.18	<12	3.25	<0.05	<6
S6	1.03	1.7	65.32	2220.5	2.82	37.28	59.45	2.48	<0.05	8.9
HP1	1.04	<1	4.32	210.4	<0.43	1.50	<12	1.13	<0.05	<6
HP2	1.03	<1	1.02	37.5	<0.43	1.72	<12	0.87	<0.05	<6
HP3	1.03	<1	9.17	488.9	0.60	1.69	<12	1.20	<0.05	<6
HP4	1.01	8.4	1.42	58.7	0.51	1.28	<12	0.91	<0.05	<6
MG1	1.00	<1	6.92	307.2	0.48	1.94	<12	<0.56	<0.05	<6
MG2	1.01	<1	19.42	1009.3	1.34	3.23	<12	0.75	<0.05	<6
MG3	1.02	<1	9.35	528.7	0.74	3.32	<12	1.25	<0.05	<6
HL1	1.00	1.5	49.02	2392.2	4.55	9.47	18.69	3.56	<0.05	<6
HL2	1.01	1.6	72.60	3264.6	4.08	9.27	21.63	1.48	<0.05	<6
HL3	1.01	3.3	24.10	1252.2	1.47	20.04	16.91	127.89	<0.05	<6
HL4	1.00	<1	166.80	16.9	1.05	3.21	<12	1.49	<0.05	<6
HL5	1.01	<1	29.29	1459.0	1.76	8.24	<12	1.78	<0.05	<6
HL6	1.01	4.0	36.21	1620.4	2.14	20.49	42.15	76.07	<0.05	<6
HL7	1.04	1.3	46.86	2483.2	3.16	8.18	<12	2.38	<0.05	<6
DT1	1.01	<1	20.05	1028.9	1.19	2.82	<12	1.20	<0.05	<6
DT2	1.01	<1	13.43	707.2	0.84	2.45	<12	1.22	<0.05	<6
DT3	1.02	<1	6.18	311.5	0.45	2.19	<12	1.71	<0.05	<6
DT4	1.00	<1	7.20	334.4	0.46	2.24	<12	1.11	<0.05	<6
DT5	1.00	<1	31.05	1465.1	1.91	5.82	<12	4.18	<0.05	<6